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Assessing Multiple Pollutant Multiple Source Cancer Risks from Urban Air Toxics: Summary of Approaches and Insights from Completed and Ongoing Urban Air Toxics Assessment Studies

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16. ABSTRACT

The primary purpose of this report is to assist State, local and other agency personnel by describing methods that have been used in assessing multiple source, multiple pollutant risks from air toxics exposures in urban areas. This report does not constitute formal EPA guidance for conducting an urban risk assessment, nor does it recommend a single approach. Instead, it identifies techniques that others have elected to employ and offers insights that may assist the reader in selecting a particular set of techniques for use in a given locale.

Major topics covered include: (1) a summary of completed and ongoing urban air toxics assessment studies, (2) ambient monitoring assessment approaches, (3) emission inventory/dispersion modeling assessment approaches (4) aspects of exposure and risk assessment, (5) control strategy evaluation, (6) data handling, and (7) evolving assessment technologies, including receptor modeling, personal monitoring and bioassay sampling.

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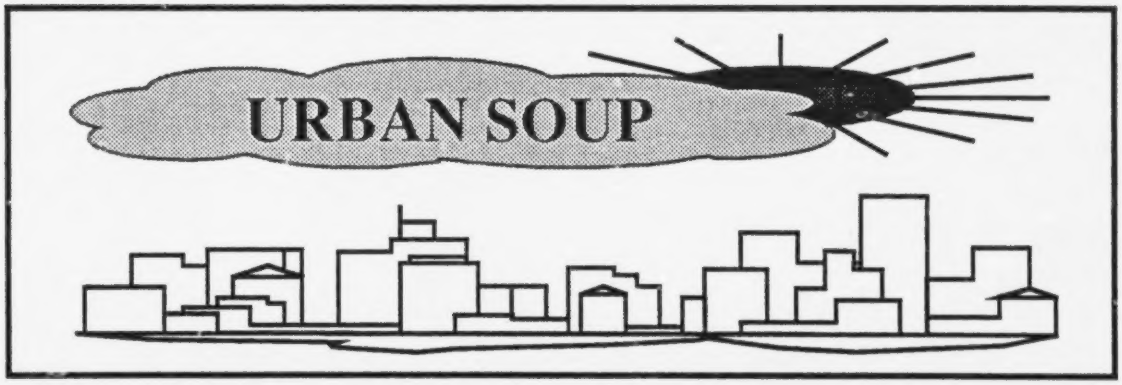

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ASSESSING MULTIPLE POLLUTANT MULTIPLE SOURCE CANCER RISKS FROM URBAN AIR TOXICS

Summary of Approaches and Insights From Completed and Ongoing Urban Air Toxics Assessment Studies

U. S. ENVIRONMENTAL PROTECTION AGENCY
Office Of Air and Radiation
Office Of Air Quality Planning And Standards
Research Triangle Park, North Carolina 27711

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EXECUTIVE SUMMARY

Background

Increasing attention is being given to the problem of air toxics exposures in urban areas. This attention initially focused on maximum exposures of individuals living near large point sources. Lately, however, some attention has shifted to the potential for widespread risks in urban communities resulting from smaller, more dispersed emitters of air toxics.

Recent Environmental Protection Agency (EPA) studies[1] suggest that multiple pollutant, multiple source exposures to air toxics may be causing 1,000 to 2,000 excess cancer cases annually in the United States and average lifetime individual cancer risks ranging from 1 in 10,000 to 1 in 1,000 (10^{-4} to 10^{-3}) in urban areas. These studies further suggest that air emissions from small point, area, and highway vehicle sources may be the largest contributors to urban air toxics cancer incidence.

Under EPA's National Air Toxics Strategy (EPA, 1985), a major activity has been the assessment of high risk urban problems. Numerous assessment efforts have been initiated in the past five years or are under way in many U.S. cities to evaluate both the nature and magnitude of the air toxics problem in urban environments. Some of these studies are examining the potential for risk mitigation through various control alternatives. EPA, in carrying out the National Air Toxics Strategy, has been encouraging urban areas to engage in these kinds of assessment activities. Moreover, EPA is promoting the coordination of State and local air toxics control programs with ozone (O_3) and particulate matter (PM_{10}) control programs to assure that, whenever possible, future State Implementation Plans (SIPs) incorporate measures that reflect co-control of air toxics. (EPA, 1987c)

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[1] Haemisegger, et al., 1985; Manale, et al., 1987; EPA, 1986; Hinman, et al., 1986; EPA, 1987a; Shikiya, et al., 1987; Shikiya, et al., 1988; EPA, 1987b; EPA, 1988.

Current Assessment Approaches—Most urban air toxics assessments to date have used either of two basic approaches: ambient air monitoring and emissions modeling. Many, in fact, have used a combination of both approaches to minimize the weaknesses of each.

The ambient air monitoring approach is conceptually simple. Measured ambient levels of air toxics in an urban area are multiplied by cancer unit risk factors to calculate individual risks. (Unit risk factors relate specific probabilities of contracting cancer to lifetime exposures to 1 $\mu g/m^3$ of a specific pollutant.) Individual risks are then multiplied by population data to calculate aggregate cancer incidence. Because of data limitations, such analyses are usually fairly crude. For example, some studies may assume that one or several measurements of each pollutant represent long-term, areawide population exposures. Ideally, ambient assessments of urban air toxics risks would be more sophisticated, involving many more ambient samples at more representative times and locations.

The second major approach involves dispersion modeling of emissions. In this approach, an emission inventory is compiled for the air toxics of concern and is modeled (using various long-term dispersion models) to predict ambient air concentrations over the urban area. These modeled ambient air concentrations are then used to estimate individual risks and aggregate incidence as if they were measured data. The dispersion modeling approach requires a comprehensive emission inventory of point, area, and highway vehicle sources.

There are advantages and disadvantages to each of these two assessment approaches. The use of reliable and representative ambient air monitoring data avoids the errors inherent in emission inventories and dispersion models, which can be considerable. In addition, the ambient air monitoring approach allows one to handle secondarily formed pollutants that are not emitted directly (e.g., photochemically formed formaldehyde) or pollutants that exist in the atmosphere resulting from gradual global buildup of background levels (e.g., carbon tetrachloride) or transport from other urban areas. (Ambient monitoring data can also be used to help verify modeling results.)

Important advantages of the dispersion modeling approach are that it allows one to predict risk reductions as a function of anticipated emission changes and it allows ambient air levels to be projected at many more receptors than may be possible in most air sampling networks. It also allows one to handle pollutants (e.g., hexavalent chromium) for which ambient sampling methods are not yet available or cost-effective. Dispersion models also allow one to estimate the impact of specific sources on particular receptors, which is not possible with ambient monitoring data.

Future Assessment Approaches—Alternative approaches are being developed for assessing the urban air toxics problem, the most notable being EPA's Integrated Air Cancer Project (IACP) and Total Exposure Assessment Monitoring (TEAM) studies. The IACP is being conducted by the EPA research labs in Research Triangle Park, North Carolina. It is a long-term interdisciplinary research program aimed at developing scientific methods and data bases to identify the major sources of carcinogenic chemicals emitted into the air or arising from atmospheric transformations. This project is performing extensive ambient air monitoring, receptor modeling, bioassays, and atmospheric transformation studies to identify the principal airborne carcinogens and, to the extent feasible, quantify the human cancer risk they may pose. Thus far, the project has concentrated on the effects of emissions from wood burning and mobile sources. Subsequent phases of this project will add other residential combustion sources (e.g., oil combustion) and industrial sources. (Lewtas, 1987)

The TEAM approach (Ott, 1986) assesses exposures by measuring pollutant levels in an individual's immediate physical environment throughout the day. In contrast to a fixed monitoring station that collects pollutants as they are carried by the ambient (or indoor) air to the location of the devices, TEAM monitoring devices are located on vests worn by individuals during their daily routine. In addition, TEAM studies measure the concentration of chemicals in an individual's drinking water and exhaled breath to determine the actual amount of the pollutant that has been taken up by the individual. Personal sampling directly addresses the variability of exposures within a population. TEAM may best contribute to future urban assessments by improving the exposure assumptions that are typically made in current assessments.

Purpose of Report

The primary purpose of this report is to assist State, local, and other agency personnel by describing methods that have been used in assessing multiple source, multiple pollutant risks from air toxics exposures in urban areas.

This report is a synthesis of assessment methods that have been made to date to quantify multi-pollutant, multi-source urban air toxics exposures and risks. It does not constitute formal EPA requirements for conducting an urban risk assessment of air toxics, nor does it recommend a single approach. Instead, it identifies techniques that others have elected to employ, and it offers insights that may assist the reader in selecting a particular set of techniques for use in a given locale. In some cases, this report discusses problems encountered in previous studies so that prospective study managers might learn some lessons. A State or local agency will have to decide from the procedures which (if any) to adapt in its particular urban assessment based on program goals, timing, and resources.

Report Organization

Chapter 1 of this report presents a brief overview of the major studies completed in the past several years that have assessed the multi-pollutant, multi-source urban air toxics problem. This overview is provided because these studies are cited throughout the body of this report. To a limited extent, the major findings of each completed study are summarized to give the reader a sense of the available evidence that suggests the existence of an urban air toxics cancer problem.

Chapters 2 through 7 describe specific activities that are common to urban air assessments. These major activities are highlighted below:

- Ambient air monitoring;

- Emission inventorying;
- Dispersion modeling;
- Exposure and risk assessment;
- Control alternative simulation and evaluation; and
- Data processing.

For each of these activities, a synopsis is given in the respective chapter of the procedures employed in the various urban assessments, and insights are offered on the suitability of these procedures in particular applications. It should be noted that not all of the above activities would necessarily be involved in a given urban assessment.

Chapter 8 summarizes various assessment methods that are currently under development and may be employed in future urban assessments. These methods, which include bioassays, receptor modeling, and personal monitoring, are being developed by EPA research laboratories. Although not the focus of this report, it is useful to be aware of these evolving techniques.

A glossary of terms relating to urban risk assessment is included in Appendix A for those not conversant with the terminology used herein. Appendix B discusses a method for representative monitoring site location relating to material discussed in Chapter 2.

Report Emphasis

Cancer Risk—Cancer has been the major focus of urban air toxics studies to date. As such, this report focuses primarily on techniques employed by various study managers to assess cancer risks. These techniques generally involve the evaluation of long-term (e.g., annual average) exposures to multiple pollutants. Techniques for evaluating specific non-cancer risks associated with short-term (acute and subchronic) exposures are only discussed briefly, reflecting the limited work done in this area. Note that this report does not deal with the development of dose-response relationships and cancer unit risk factors. It is assumed that potency data for different pollutants are available from other sources.

Measures of Cancer Risk—Two common measures of excess cancer are incidence and individual risk. Excess cancer incidence is a measure of the number of excess cancer cases associated with air toxics exposures within an urban area, whereas individual risk is the probability that an individual will contract cancer from a particular exposure.

There are many variations of each of these measures. For example, incidence can be additive, covering multiple pollutants. Incidence can reflect areawide cancer cases or can be specific to particular "hotspots," such as neighborhoods or industrial parks. Incidence can reflect the number of cases expected in a single year or within a 70-year "lifetime." Incidence is sometimes population-normalized, for example, adjusted per million population. A commonly used measure in multi-pollutant, multi-source urban air toxics assessments is annual, areawide excess cancer incidence.

In the same way, individual risk can refer to the cancer probability associated with multiple or single pollutant exposures, the probability associated with the average exposure within a broad area or with the exposure at a particular receptor, or the probability of contracting cancer in a single year or during a 70-year lifetime.

Screening Assessments—Most of the urban assessments completed to date are best described as screening (or scoping) studies, performed to yield an order-of-magnitude estimate of the relative nature of the urban cancer problem, rather than to provide absolute predictions of incidence and individual risks. Screening studies are often considered more credible for prioritizing subsequent assessment efforts than for defining direct regulatory measures or for predicting risks to specific individuals by specific sources and pollutants. If, for example, the screening analyses described herein would point to a single point source or a cluster of sources as posing particularly high risks, more detailed assessment techniques might be needed, especially in situations (e.g., permit review or standard setting) where specific regulatory actions were contemplated.

Risk Assessment vs. Risk Management—Quantitative risk assessment is a process by which a factual base of information is used to estimate the probability of incurring some risk (e.g., cancer) due to exposure to a specific chemical or chemical mixture. In contrast, risk management is the decision-making process by which some action is taken or some policy is formed concerning a potential risk to the environment and/or to human health. Risk management differs from risk assessment in that management of risk usually considers political, economic, and social issues in the decision-making process. This document mainly deals with the methods to assess urban air toxics risks, including the projection of risk reductions associated with control alternatives.

The Changing Nature of Urban Assessments

There are myriad techniques, assumptions, and data involved in urban assessments, many of which may change in the next few years. Cancer unit risk factors, for example, are subject to significant, often order-of-magnitude, changes as new dose-response data become available. Ambient air monitoring and source sampling techniques for air toxics are evolving, allowing certain compounds to be measured accurately at levels not previously possible. Techniques are also becoming available to evaluate mutagenicity of specific urban air compounds as they age and transform photochemically.

It is important to be aware of the changing nature of urban assessments for two reasons. First, it means that no two completed assessments have likely used the same procedures, have made the same assumptions, or have used the same data in estimating urban risk. If one examines the emission factors, unit risk factors, modeling assumptions, pollutant coverage, etc., in the various studies, significant divergence is often apparent. Hence, care needs to be taken when relating results from one study to another to avoid "apples and oranges" comparisons.

Second, the information presented in the following chapters will undoubtedly change as the perception of the urban problem matures and as more assessment options become available. Moreover, the underlying emission inventory and ambient air quality data bases upon which such assessments are based, should improve with time. Thus, those responsible for urban risk assessments should keep abreast of these changes and reflect them as much as possible.

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CHAPTER 1

AN INTRODUCTION TO THE URBAN AIR TOXICS STUDIES

The following subjects are covered in this chapter:

- Defining the urban air toxics problem
- Summary of completed and ongoing urban air toxics assessments
- Other urban air toxics assessment activities

1.1 Defining the Urban Air Toxics Problem

The perception has been evolving over the last few years that an air toxics "soup" exists over cities that is imposing an uncertain, but possibly significant, public health risk of increased cancer. This concern has been described by the following terms:

- Urban hot spot problem
- Multiple source, multiple pollutant problem
- High-risk urban problem
- Urban soup

Whatever term one uses to describe the phenomenon, the concerns are basically the same:

- That traditional air toxics programs, which typically emphasize major point sources and single pollutant analyses, may not be addressing the most significant cancer risks in urban areas;
- That criteria pollutant control programs, which are known to effect some air toxics reductions through indirect means, may not be sufficiently addressing certain air toxics problems;

- That small point and area sources and road vehicles may be causing more of a problem than has been recognized heretofor in air toxics programs;
- That simultaneous exposures to urban air toxics mixtures from many sources can lead to significant aggregate risks in urban areas, risks that are not being realistically assessed by current pollutant-by-pollutant risk assessments; and
- That byproducts of irradiated and aged urban air mixtures may be much more potent than "fresh" (i.e., point of release) emissions traditionally analyzed in some risk assessments.

Thus, whether one calls the problem "urban soup" or something else, the point of conducting an urban air toxics assessment is the same—to shed light on these concerns and determine whether additional air toxics controls are needed.

1.2 Summary of the Completed and Ongoing Urban Air Toxics Assessments

This report is a compilation of procedures that are being employed for conducting multi-pollutant, multi-source urban air toxics assessments. The procedures described in this report are derived from a number of urban studies either completed in the last several years or currently under way.

The purpose of this chapter is to introduce the reader to the studies alluded to throughout the remainder of this report. Hence, what follows is a brief discussion of the purpose and approach of each study, along with an overview of its major findings. As a rule, a short descriptive title is assigned to each study (e.g., the "Six Months Study") so that hereafter, for ease of reference, the full title of each assessment need not be repeated.

The reader is cautioned that all of the risk and incidence values cited herein, while appearing very precise and certain in some cases, are acknowledged to be very preliminary and tentative by the respective study authors and by the authors of this report. The many uncertainties, assumptions, and limitations inherent in the procedures used in these studies are documented in the remainder of this report, and should be kept in mind before using any of the following data and conclusions.

The Six Months Study

This study is formally entitled The Air Toxics Problem in the United States: An Analysis of Cancer Risks for Selected Pollutants. (Haemisegger, 1985) It represents EPA's first comprehensive analysis of the air toxics problem and provides a basis for the Agency's National Air Toxics Strategy. (EPA, 1985) The Six Months Study is regarded as a "scoping" study only, useful in a relative sense to yield rough approximations of total incidence and individual risks.

Three major analyses were undertaken as part of the Six Months Study to estimate cancer incidence and individual lifetime risks. An "Ambient Air Quality Study" used ambient data for 5 metals, 10 volatile organic compounds, and benzo(a)pyrene [B(a)P] to assess risks. Two other analyses—a "NESHAP Study" and a "35-County Study"—used emission estimates and exposure models to estimate incidence and maximum individual risks associated with the pollutants selected. In these analyses, a "B(a)P surrogate" approach was used to estimate cancer incidence associated with products of incomplete combustion (PIC). To do this, a dose-response coefficient relating lung cancer incidence and PIC was generated from epidemiological studies, and cancer incidence associated with PIC exposure was estimated by applying this dose-response coefficient to B(a)P levels. Finally, quantitative risk assessments available from other EPA activities for asbestos, radionuclides, and gasoline marketing were incorporated into the study.

The Six Months Study shows that both point and area sources contribute significantly to the air toxics problem. Large point sources are associated with high maximum individual lifetime risks (commonly 10^{-4} to 10^{-2}), whereas the additive lifetime individual risk due to simultaneous exposure to 10 to 15 pollutants range from 10^{-4} to 10^{-3}. The latter risks do not appear related to specific point sources, but instead, are associated with the complex mixtures typical of urban ambient air. For the pollutants examined, additive cancer incidence averaged about 6 excess cases per million people per year.

This study very tentatively suggests that EPA's criteria pollutant programs have significantly reduced air toxics levels. An analysis of 16 pollutants was completed using both monitoring and emission data in order to evaluate progress made on air toxics between 1970 and 1980. The estimated cancer incidence rate for these air pollutants in 1980 was less than half that for 1970.

The Integrated Environmental Management Project (IEMP)

Under the Integrated Environmental Management Project (IEMP), EPA's Office of Policy, Planning, and Evaluation (OPPE) conducted a number of urban (or "geographic") studies to evaluate the multi-media aspects of various environmental issues, with general emphasis on toxics exposures from drinking water, hazardous wastes, and air. The geographic areas covered by the IEMP studies to date are Philadelphia, Pennsylvania (EPA, 1986); Baltimore, Maryland (Manale, 1987); Santa Clara ("Silicon Valley"), California (Hinman, 1986); and Kanawha Valley, West Virginia (EPA, 1987a). Another IEMP is under way in Denver, Colorado, at this writing.

The concept of integrated environmental management developed out of EPA's recognition that there are potential drawbacks to the traditional approaches for developing environmental regulations. Agencies have traditionally focused on individual industries, pollutants, and media—thereby potentially limiting their ability to determine where, among the various media, their resources could best be employed to optimize health protection. Moreover, the traditional approach may not ensure that pollution controls are not merely shifting risk from one medium to another. Thus, comparing the different media risks to help set priorities allows environmental managers to focus limited resources in a manner that will achieve the greatest public benefit.

The cancer assessment approaches used in each of the four IEMP studies completed to date are fundamentally similar. First, each study modeled emissions data to project ambient air concentrations. These projected concentrations were then distributed over a population grid to estimate exposures, which, in turn, were multiplied by unit risk factors to estimate individual cancer risks and incidence. Some ambient air measurements were made as part of these studies to complement the modeling results and to help check for errors and omissions in the inventory.

The results of the IEMPs generally support the conclusions of the Six Months Study. Excess cancer incidence from exposures to 10 to 20 toxic air pollutants ranges from about 1 to 18 cases per year per million persons. The relative contributions of point and area sources vary, with area sources generally predominant (except in Kanawha Valley with its numerous, large chemical complexes). A combination of metals (e.g., chromium), polycyclic organic matter (e.g., B(a)P), and toxic volatiles (e.g., benzene) account for most of the excess urban cancer incidence. Additive individual lifetime cancer risks to the most exposed individuals around point sources vary considerably, ranging from 10^{-4} to as high as 10^{-2}. Average individual lifetime risks of 10^{-4} are common in many areas not impacted by particular point sources.

Several of the IEMPs attempted to evaluate noncancer risks. The only pollutant present at levels associated with noncancer health effects was benzene, both around a large steel manufacturing complex and near major traffic intersections. One IEMP study also suggested that ambient xylene concentrations near large wastewater treatment plants may have the potential for causing noncancer health effects.

Clark County Study

In early 1985, after reviewing EPA's Six Month Study, the Clark County Health District attempted to address the magnitude and nature of the air toxics problem in Las Vegas, Nevada, using a simple ambient monitoring approach. (EPA, 1987b) The District initially estimated average, annual Valley-wide levels for 11 chemical substances, based on short-term data, and multiplied these levels by unit risk factors to arrive at an additive cancer incidence. The District had some knowledge of the emissions and/or air quality levels of the 11 substances in the Las Vegas Valley. Most air quality measurements were obtained from an established station in east central Las Vegas (which has a history of high CO and TSP levels) and from a station in the southeast Las Vegas Valley (which has a history of high ozone levels and complaints of chlorine odor and eye irritation). The toxics data were derived from the District's efforts to characterize urban haze and to develop specific hydrocarbon profiles to support ozone modeling exercises. The urban haze research provided short-term data for various metals and carbonaceous material.

Results of the initial screening compilation show that products of incomplete combustion (PIC), asbestos, benzene, perchloroethylene, and chromium account for most of the cancer incidence from air toxics exposures in the Valley. Following the initial screening, the District then refined its estimates of annual averages for particular substances of interest and recalculated at the annual Valley-wide cancer incidence. Totaling the chemicals, the District calculated that the annual incidence is 4.3 to 9.4 cancers per million people per year. PIC is the overwhelming contributor to total incidence from air toxics generated almost equally by three source categories: (1) cars using leaded gasoline, (2) diesel trucks and buses, and (3) wood burning fireplaces. Other pollutants of lesser importance include arsenic, benzene, cadmium, and chromium.

South Coast—MATES

The South Coast Air Quality Management District (SCAQMD) has conducted one of the most comprehensive studies to date of the urban air toxics problem. This study, termed the Multiple Air Toxics Exposure Study (or "MATES"), is a multiple year evaluation of community exposure to air toxics in the Los Angeles metropolitan area. MATES has been the focal point for urban air toxics study by SCAQMD since 1985; however, various other independent activities that have been ongoing have provided data for MATES. These other activities include certain monitoring and emission inventory activities, an "in-vehicle" characterization of exposures to commuters inside their vehicles, and the development of a dispersion/exposure/risk model named "SCREAM" (South Coast Risk and Exposure Assessment Model). SCREAM is an enhanced version of EPA's Human Exposure Model (HEM). (Barcikowski, 1988)

The MATES project is documented in a series of working papers, each reflecting major components of activity. This series includes, in order:

- A quality assurance plan (SCAQMD, 1985)
- A monitoring site selection plan (SCAQMD, 1987a)
- An emissions data summary (SCAQMD, 1987b)
- A modeling results summary (SCAQMD, 1987c)

- A monitoring results summary (SCAQMD, 1988).

The MATES project utilized data from two different ambient sampling efforts. One set of data came from ongoing sampling activities at existing SCAQMD sampling sites. The second effort was carried out by SCAQMD as part of MATES and involved setting up ten new/temporary sampling sites in areas suspected of having elevated concentrations, based on dispersion modeling of emissions data and various on-site criteria.

Risk estimations at the sampling locations indicate that there are areas within the South Coast Basin where the combined impact of multiple air toxics are significantly higher than the Basin average. The combined individual risks from simultaneous exposure to the 20 pollutants under study (14 organic gases and 6 metals) ranged from about 1 in 10,000 to 1 in 1,000, predominantly due to benzene and chromium. Higher concentrations of most of the pollutants were measured in the winter than in the summer because of such meteorological conditions during the winter months as low ambient temperatures and longer hours of surface inversions leading to reduced vertical mixing. (Barcikowski, 1988)

The modeling effort in MATES used, as input, emission inventories of point, area, and highway vehicle emissions compiled in 1982 and updated to 1984. The estimation of population exposures to one or more air toxics was conducted by first using dispersion modeling of emissions data to calculate the long-term concentrations at centroids of census areas and then multiplying the calculated concentrations with the population that each centroid represents. The areawide risks, in terms of incremental cancer cases, were then calculated by multiplying the population exposure with the chemical-specific unit risk factors developed by the California Department of Health Services. A linear relationship was assumed and the exposure/risks associated with multiple sources and with species of air toxics are assumed additive. SCREAM was used to apportion the number of excess cancer cases by source category and by pollutant, and to identify high-risk chemical substance and source categories. SCREAM can also be used to identify high-risk locations and to estimate control measure effectiveness in reducing exposure, cancer risk, and number of cases.

Of the 20 air toxics studied in the emission inventory/dispersion modeling portion of MATES, benzene and hexavalent chromium have the greatest impact on the Basin's population. Almost the entire population is exposed to ambient benzene and hexavalent chromium concentrations corresponding to an upper-bound risk of 1 in 10,000 or higher. As an upper-bound estimate, this portion of the study found that about 160 excess cancer cases would result annually in the Basin due to the combined exposure to all 20 pollutants, or about 15 excess annual cases per million population. (Barcikowski, 1988; Shikiya, 1987c)

Subsequent activities include the enhancement of SCREAM and the development of various risk mitigation strategies. The SCREAM computer model is being enhanced and will include options for population mobility, indoor-versus-outdoor exposures, and noninhalation routes of exposure. SCREAM will also be enhanced to provide better estimates of individual risk and community cancer burden in regions and subregions of the Basin.

The two major toxic air contaminants of concern, benzene and hexavalent chromium, will be reduced through control efforts at the local District level and by programs at the State level. SCAQMD has recently adopted a measure to control emissions from chrome platers and is moving forward with rulemaking to control hexavalent chromium emissions from cooling towers. Options for reducing benzene contents in motor vehicle fuels are being evaluated. More stringent requirements on motor vehicles that are aimed at reducing hydrocarbon emissions will also reduce emissions of benzene from motor vehicles. (Barcikowski, 1988)

5 City Controllability Study

This study (EPA, 1988a), currently under way, is modeling ambient exposures to 25 air toxic compounds in five major urban areas. It also estimates how exposure levels are likely to change in the future as a result of alternative control scenarios. The urban areas were chosen to represent a cross-section of potential problem types and to ensure geographical diversity. Air toxics exposures in the five study areas were estimated via EPA's Human Exposure Model (HEM). A 1980 baseline emissions inventory was established to quantify emissions data for input into the model. Emissions data were developed by various techniques, building on the source data in EPA's National Emissions Data System (NEDS) and special State and local inventories. Various 1995 projection inventories, reflecting expected and alternative control scenarios, will be simulated to evaluate the effectiveness of various mitigation options. Risks from secondary (i.e., photochemically formed) formaldehyde were estimated by superimposing measured concentrations over the modeling domain. Polycyclic Organic Matter (POM) risks were estimated using "comparative potency factors" for individual source categories rather than the "B(a)P surrogate approach" used in the Six Months Study. Comparative potency factors are being developed in EPA's Integrated Air Cancer Program (IACP) and relate the potency of various organic mixtures to known carcinogens by comparing their respective mutagenicities. (Lewtas, 1987)

The primary measure of risk in this study is aggregate cancer incidence. Preliminary results suggest that cancer incidence in the five cities ranges from about 2 to 10 excess cases annually per million persons, in general agreement with other urban assessments. Urbanwide individual lifetime risks range from about 1.5×10^{-4} to 7×10^{-4}. The major pollutants of concern are polycyclic organic matter (POM), hexavalent chromium, 1,3-butadiene, formaldehyde, and benzene. Small point and area sources contribute most to total incidence, with road vehicles, cooling towers, chromium platers, solvent use, and wood combustion being predominant.

An important finding of this study is that considerable air toxics control will be achieved by 1995 from measures reasonably anticipated to be in effect because of State Implementation Plans (SIPs), New Source Review (NSR), the Federal Motor Vehicle Control Program (FMVCP), and National Emission Standards for Hazardous Air Pollutants (NESHAPs). As suggested by the Six Months Study, some of this reduction is indirectly due to criteria pollutant measures rather than to direct controls on air toxics.

Motor Vehicle Study

This study, entitled Air Toxics Emissions from Motor Vehicles, (Carey, 1987), focused primarily on cancer risks posed by road vehicle emissions in the United States. It is not an urban assessment *per se*, but the results are useful in an urban soup context as most vehicular travel is in urban areas. The report considered all vehicular air emissions for which EPA has cancer unit risk estimates. Specific pollutants and pollutant categories included are diesel particulate, formaldehyde, benzene, gasoline vapors, gas phase organics, organics associated with non-diesel particulate, dioxins, asbestos, vehicular interior emissions, and metals.

Several unique aspects to this study are worth highlighting. First, an activity pattern model was used to reflect changing exposures to sets of population groups as they change location from one microenvironment to another during their day-to-day activities. These microenvironments included street canyons, parking garages, and indoor air, as well as various urban and suburban settings. This approach for estimating exposures differs markedly from all other urban assessments to date. Second, this study included an approach to evaluate risk from polycyclic organic matter (POM) that used a comparative potency method. In this method, the potency of organic extracts of diesel particulate and gasoline particle organics was compared with the potencies of extracts from sources for which epidemiological data were available (Albert, 1983). This contrasts with the so-called "B(a)P surrogate approach" used in EPA's Six Months Study wherein B(a)P was used as a surrogate for all POM.

This study associated 385 to 2,286 annual excess cancer cases nationwide with air toxics emissions from road vehicles. From 2 to 11 excess cancer cases are predicted per year per million people in urban settings. This incidence drops roughly 40 percent by 1995 because of more stringent diesel particulate standards for both light and heavy duty vehicles and because of the increasing use of three-way catalyst equipped vehicles coupled with the phaseout of non-catalyst equipped vehicles. The pollutants contributing most to total cancer incidence are, in rough order of importance, diesel particulate, 1,3-butadiene, benzene, gasoline particulate-associated organics, formaldehyde, and asbestos. The bulk of the formaldehyde risk is due to secondary (i.e., photochemically produced) formaldehyde.

Integrated Air Cancer Project (IACP)

The Integrated Air Cancer Project (IACP) is being conducted by the EPA research labs in Research Triangle Park, North Carolina. It is an interdisciplinary research program (Lewtas, 1987) aimed at developing scientific methods and data bases to identify the major sources of carcinogenic chemicals emitted into the air or arising from atmospheric transformations. This project is performing extensive ambient air monitoring, receptor modeling, bioassaying, and atmospheric transformation studies. Thus far, the study has concentrated on the effects of emissions from wood burning and mobile sources. Subsequent phases of this project will add other residential combustion sources (e.g., oil combustion) and industrial sources.

The major long-term goals of the project are as follows:

1. To identify the principal airborne carcinogens.
2. To determine which emissions sources are the major contributors of carcinogens to ambient air. Major carcinogenic emissions sources will be determined by field studies with simultaneous emission characterizations and ambient monitoring, followed by source apportionment calculations.
3. To improve estimates of comparative human cancer risk from specific air pollution emission sources. The study will develop a comparative methodology to evaluate and apply short-term mutagenesis and animal carcinogenesis data on emission sources. Improved human exposure estimates will be developed for complex emission products and individual carcinogens, including transformation products.

Field studies have been conducted to date in Raleigh, North Carolina, Albuquerque, New Mexico, and Boise, Idaho, with plans to add Roanoke, Virginia. Most of the work that has been reported has been in Raleigh and Albuquerque, and has focused mainly on methods development. The Boise results are not available as of this writing.

An important result has been the observation of elevated mutagenicities of irradiated and aged gas phase mixtures. The mutagenicities of wood smoke, auto exhaust, and several common industrial organics (toluene, propylene, and acetaldehyde) all increase by an order-of-magnitude or more after the mixtures are irradiated and allowed to react for several hours. These results suggest that the transformations of complex mixtures may contribute significantly to the total burden of mutagens or carcinogens in the environment and should be considered in assessing risk. (Shepson, 1987)

TEAM (Total Exposure Assessment Methodology) Studies

Current urban assessments traditionally assume that the individual's entire exposure to air toxics is a result of continuous exposure to outdoor air. The simplistic assumptions and procedures involved in most current assessments, therefore, do not realistically approximate actual personal risks from other routes of exposure, such as from indoor air, workplace exposures, automobile interiors, etc.

In contrast, EPA's TEAM studies (Ott, 1986; Ott, 1988) have been designed to test statistical and chemical methodologies for estimating total human exposure across different microenvironments and various media. This research program has sought to establish, for each of 20 or so organic chemicals, the relative importance of certain routes of exposure (air, water) and whether predictable correlations exist between exposure and body burden (as measured through analysis of breath).

The TEAM approach assesses exposures by measuring the amounts of pollutants in an individual's immediate physical environment throughout the day. In contrast to a fixed monitoring station that collects pollutants as they are carried by the ambient (or indoor) air to the location of the devices, TEAM monitoring devices are located on vests worn by individuals throughout their daily routine. In addition, TEAM measures the concentration of chemicals in an individual's drinking water and exhaled breath in order to determine the actual amount of the pollutant that has been taken up by the individual. Personal sampling directly addresses the variability of exposures within a population and should help to improve the exposure assumptions that are typically made in urban assessments.

Since 1979, the TEAM studies have measured the personal exposures of hundreds of people across the United States to 20 to 26 volatile organic compounds (VOCs) in air and drinking water. The chemicals were selected on the basis of their toxicity, carcinogenicity, mutagenicity, production volume, presence in the air or drinking water at the field sites, existence of National Bureau of Standards (NBS) permeation standards, and amenability to collection on the sorbent Tenax. The following cities are among those that have been or are being evaluated: Bayonne-Elizabeth, New Jersey; Greensboro, North Carolina; Devil's Lake, North Dakota; the California cities of Los Angeles and Antioch; Pittsburgh, Pennsylvania; and Baltimore, Maryland. The latter application was coupled directly with the Baltimore IEMP in order to evaluate the appropriateness of the use of ambient monitoring and modeling data as proxies for total individual exposures in risk assessments. The interpretation of the Baltimore TEAM results is still in progress at this writing. Results from the TEAM studies "indicate that even in major chemical manufacturing and petroleum refining areas such as Los Angeles and northern New Jersey, personal exposures to many toxic and carcinogenic VOCs exceeded outdoor concentrations by 100 to 400 percent. Breath measurements of the 550 residents who took part in the study were significantly correlated with the preceding air exposures. It was concluded that personal activities such as smoking, using room air deodorizers, wearing dry-cleaned clothes, and even using hot water were responsible for a major portion of most people's exposure to benzene, para-dichlorobenzene, tetrachloroethylene, and chloroform, respectively." (Wallace, 1988) Prior TEAM studies (Ott, 1985) have shown that levels of 11 important organic compounds, some of which are regarded as potential carcinogens, are found to be significantly higher indoors than outdoors. These chemicals included chloroform, 1,1,1-trichloroethane, benzene, carbon tetrachloride, trichloroethylene, perchloroethylene, styrene, meta- and para-dichlorobenzene, ethylbenzene, and xylene isomers. The sources appear to be inside the home, probably in furniture, paint, solvents, drapes, carpets, spray cans, clothing, and construction materials.

In addition to the above described method, which directly measures daily exposure profiles for a representative cross-section of population within an area, TEAM is also constructing estimates of personal exposure profiles by combining information on human activities with microenvironmental field data. In essence, concentration is measured in selected microenvironments in which a person will be exposed throughout a day. Then, an integrated exposure is computed as the product of (1) the concentration of each pollutant encountered in each microenvironment and (2) the time the person spends there, divided by the total time of exposure. For VOCs, typical microenvironments may include gas stations, dry-cleaning stores, freshly painted rooms, and households where solvents are stored indoors. The construction of time-activity exposure profiles allows the extrapolation of the results to larger populations and other locales, using models that incorporate human activity patterns. (Ott, 1985).

Staten Island/New Jersey Study

The Staten Island/New Jersey study (EPA, 1986b), begun in 1986, was initiated to provide data on the scope and magnitude of the urban air toxics problems in the Staten Island/New Jersey (Middlesex and Union Counties) areas. This study will rely primarily on ambient monitoring to characterize air quality in the study area for selected toxic pollutants. The measured data will be used to estimate exposures and to understand better the interrelationship between indoor and ambient exposures. Plans are for emission inventory data to be compiled to formulate hypotheses linking major contaminants to potential sources and to evaluate general abatement strategies.

The Staten Island area was selected for intensive study based on the high density of industry in the area, a long-standing history of odor complaints leading to a perception that an air toxics problem exists, and the high quality of technical expertise available from State and local organizations to study the situation. As of this writing, no results have yet been reported.

Southeast Chicago Study

The Southeast Chicago study (Summerhays, 1987) was initiated by EPA as a result of citizen concerns regarding the environmental safety of the heavily industrialized area in the vicinity of Lake Calumet, located in southeast Chicago, Illinois. A broad range of environmental concerns was raised relating to air, water, and land pollution issues. As a response, an emission modeling study was undertaken by EPA's Region V office to ascertain whether air toxics exposures may be causing significant cancer risks.. The initial step was the development of an air toxics emission inventory over a broad area of Chicago, encompassing the receptor area of concern. The inventory included 47 suspected carcinogens—22 nonhalogenated compounds, 17 halogenated compounds, and 8 inorganic species (mostly metals). As part of the inventory process, questionnaires were sent to 29 of the 88 point sources in the area.

Dispersion modeling using ISC (for point sources) and CDM (for area sources) was performed to estimate ambient concentrations of air toxics in the receptor grid. The resulting concentrations were applied to population data for the same receptor grid and multiplied by cancer unit risk factors to estimate individual risk and aggregate incidence.

No results are available from this modeling study as of this writing.

Urban Air Toxics Monitoring Program

In 1987, EPA initiated a screening program entitled the Urban Air Toxics Monitoring Program (EPA, 1987c). The primary purpose of the program is to collect air quality data to support State and local agency efforts to assess the nature and magnitude of the urban air toxics problem in their respective areas. In 1987, 19 State and local agencies agreed to participate in this program. Depending on future funding allocation and interest shown by State and local agencies, this program may be continued for several years.

EPA's objectives in promoting and supporting this monitoring program are:

- To provide estimates of annual concentrations of selected air toxics;
- To provide information for prioritizing and planning future work and sampling on a more in-depth and pollutant-specific basis in local areas;
- To provide a means to identify prevailing pollutants and possible source types that may need further assessment; and
- To identify a means to evaluate and prioritize future air toxics mitigation programs.

The program calls for State and local agency personnel to collect ambient air samples for subsequent analysis either by EPA or an EPA contractor for specific toxic compounds. Under EPA direction, a central contractor will bring the necessary sampling train to each site, assemble the apparatus, instruct the sampling technician on equipment operation and maintenance, and provide detailed sample shipping information. Participating personnel will schedule a meeting date and location for the contractor to set up and demonstrate the sampling trains.

Three different types of ambient air samples are collected. The first set of air toxics samples is collected in stainless steel canisters for 24-hour periods every 12 days for one year. After sample collection, the canisters are air shipped to a central laboratory for analysis. The samples are analyzed for about 30 volatile organic compounds by gas chromatography equipped with multi-detector capability. The second set of samples is collected in cartridges for determination of formaldehyde and other aldehydes. The third set is total suspended particulate (TSP) matter collected from a high-volume air sampler for determination of about 14 metals and B(a)P.

No results are available as of this writing.

Bay Area Toxics Monitoring Study (BAAQMD, 1987)

The Bay Area Air Quality Management District (BAAQMD), in conjunction with the California Air Resources Board, conducts ambient sampling for 11 volatile air toxics at a network of 15 monitoring sites in the Bay Area, comprising one of the largest urban air toxics networks operating in the United States. The network is oriented to population exposure, with a background site north of San Francisco near the ocean. The pollutants analyzed are believed by the Bay Area AQMD to represent a large part of the risk of volatile toxic substances to the general population. Twenty-four hour samples are taken in Tedlar bags and are analyzed by gas chromatography. Sampling frequency is twice per month per site.

Measurements were started in 1986 and are ongoing. Results to date suggest the benzene levels constitute the majority of risk associated with the measured exposures and are largely derived from vehicular activity.

1.3 Other Urban Air Toxics Assessment Activities

EPA has been encouraging State and local air pollution control agencies to assess their urban air toxics problems. Numerous assessment activities have been undertaken in the past several years in the areas of ambient air monitoring, emissions inventorying, exposure and risk characterization, and mitigation analysis. Figure 1-1 shows those U.S. cities where some type of assessment activities have been initiated involving one or more of these activities. Not all of these activities will lead to multi-pollutant, multi-source cancer risk assessments. Table 1-1 lists the specific activities undertaken in 30 urban areas having populations exceeding one million persons.

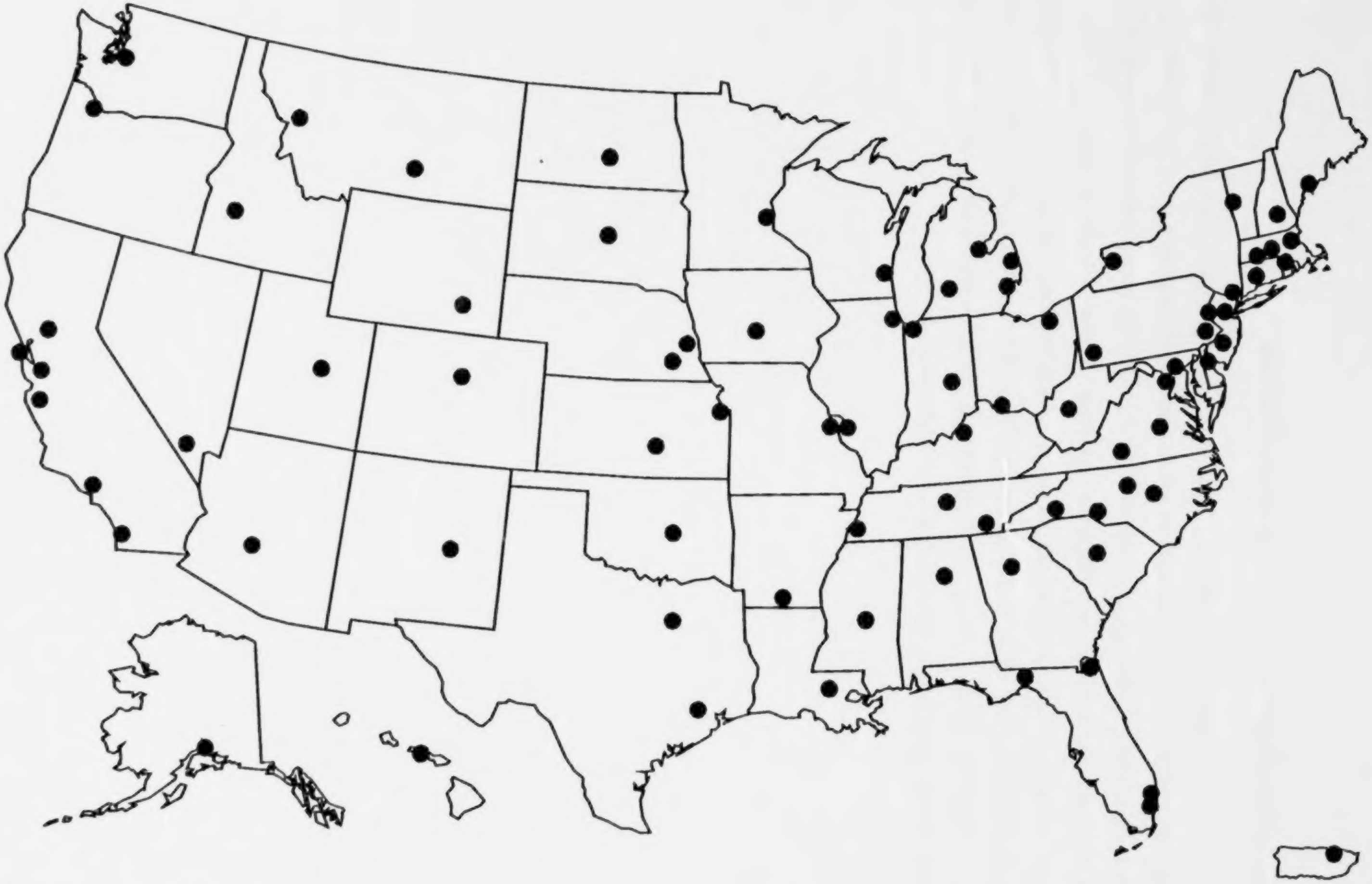

Figure 1-1 Urban Areas With Air Toxics Assessment Activities in 1988

TABLE 1-1

URBAN AIR TOXICS - ACTIVITY IN SELECTED URBAN AREAS - NOVEMBER 1988

EPA REGION	STATES	CITIES OR AREAS	Planned, ongoing or recently completed activities			
			Air Toxics Monitoring	Emissions Inventory	Risk Assessment	Mitigation Analysis
I	MA	Boston	✓			
II	NY	Staten Island	✓	✓	✓	
		Buffalo	✓			
	NJ	Northeast NJ (near S.I.)	✓	✓	✓	✓
III	MD	Baltimore	✓	✓	✓	✓
	PA	Philadelphia	✓	✓	✓	✓
		Pittsburgh	✓	✓	✓	
	DC	Washington	✓	✓		
IV	GA	Atlanta	✓	✓	✓	
	FL	Miami	✓	✓		
		Jacksonville	✓			
V	IL	Chicago	✓	✓	✓	✓
	MI	Detroit	✓			
	MN	Minneapolis	✓	✓	✓	
	OH	Cleveland	✓	✓	✓	
		Cincinnati	✓	✓	✓	
	WI	Milwaukee		✓		✓
VI	TX	Dallas-Fort Worth	✓	✓	✓	
		Houston	✓	✓	✓	
	LA	Baton Rouge	✓	✓		
VII	MO	St. Louis				
		Witchita	✓	✓		
VIII	CO	Denver	✓	✓	✓	✓
IX	AZ	Phoenix		✓		
	CA	Los Angeles	✓	✓	✓	✓
		San Diego	✓	✓	✓	✓
		San Francisco	✓	✓	✓	✓
		San Jose	✓	✓	✓	✓
X	WA	Seattle	✓	✓	✓	✓
	OR	Portland	✓	✓	✓	

Source: Lamason, 1988

Chapter 1

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CHAPTER 2

AMBIENT AIR QUALITY MONITORING

The following subjects are covered in this chapter:

- Use of ambient air monitoring data in urban assessments
- Pollutant coverage
- Site selection
- Sampling periods, frequencies, and duration
- Sampling and analytical techniques
- Evolving monitoring techniques
- Insights into the use of monitoring in air toxics programs

2.1 Use of Ambient Air Monitoring Data in Multi-Pollutant, Multi-Source Urban Assessments

Multi-pollutant, multi-source urban air toxics assessments commonly use ambient air monitoring data to help characterize exposures and risks related to toxic air pollutants. Ambient air data are most often used in conjunction with dispersion modeling of emissions data to minimize the limitations inherent in using either type of data alone.

Estimating exposures using data from ambient air monitoring programs in the studies under review has been conceptually straightforward. Ambient levels of air toxics have been measured over urban areas to yield estimates of population exposures. These exposure levels were then multiplied by the respective potencies (i.e., cancer unit risk factors) of each pollutant to estimate individual cancer risks, which, in turn, are superimposed over population distributions to estimate excess cancer cases. Such analyses are often fairly crude because only a comparatively few measurements of each pollutant are available to represent long-term, area-wide population exposures. Fortunately, the increasing emphasis on ambient air toxics monitoring by States and local agencies is providing more complete data for exposure and risk assessments.

There are important advantages in using ambient air monitoring data as part of urban assessments, the principal one being the ability to directly characterize concentrations at key receptor sites of interest. In this regard, the use of reliable and representative ambient air monitoring data, instead of modeled concentrations based on emissions data, circumvents the errors inherent in emission inventories and dispersion models, which can be considerable. In fact, some pollutants that may be covered by ambient air quality monitoring programs may not be included in emissions inventories. In addition, ambient air quality monitoring allows one to evaluate pollutants that are not emitted directly, such as photochemically formed formaldehyde or peroxyacetyl nitrate (PAN) or other transformation products, or to measure compounds that exist in the atmosphere because of gradual build-up of background levels (e.g., carbon tetrachloride) or transport from other urban areas. Finally, ambient air measurements provide a direct reference from which to assess dispersion model performance and emission inventory accuracy.

Ambient air monitoring data were used in many of these studies for purposes in addition to multi-source, multi-pollutant cancer risk assessments. For example, ambient air measurements were used in the Philadelphia and Kanawha Valley studies to help identify maximum individual exposures in the vicinity of major point sources. This might also be done to help verify that the conditions in an operating permit are being met. Ambient air measurements can also be used to assess the adequacy of an emission inventory and/or dispersion model are used to predict ambient air concentrations. Poor agreement of measured and predicted ambient concentrations may reflect inventory error (e.g., a major overlooked source) and/or poor model performance. Finally, ambient monitoring data are now being used for mutagenicity bioassays and source apportionments. Regulatory agencies may want to consider multiple program uses of any measurements that are taken as part of an urban soup assessment to optimize the use of its ambient monitoring resources.

The major disadvantages of ambient air monitoring are the high costs and uncertainties involved with measuring low concentrations of specific air toxics. Obtaining a sufficient number of samples to show variations in concentration in time and space can be prohibitively expensive. Also, high detection limits, sample contamination (in the field, in transit, and in the laboratory), uncertain identification of target compounds, and other sources of error have hampered many data collection efforts; these should be anticipated and carefully evaluated during the design stages of monitoring programs. Finally, ambient sampling and/or analytical procedures are not yet proven for all pollutants (e.g., hexavalent chromium, 1,3 butadiene, acrylonitrile, and ethylene oxide). All of these disadvantages suggest that EPA and State and local agencies must optimize the number of sampling sites and measurements in any assessment, and must coordinate with other applied programs and the research community before initiating a major sampling effort.

In the Evolving Monitoring Technologies section of this chapter, several promising approaches are described to reduce the cost of monitoring programs for toxic air pollutants. Techniques such as sample compositing, sampling over longer time periods, and periodic expansion to a larger sampling network may dramatically increase the information provided per monitoring dollar spent.

2.2. Decisions Affecting Monitoring Plan Development

Design of an effective monitoring plan is the most important step in the development of a monitoring program. The two basic elements of concern are (1) clearly specified data quality objectives (definition of the type, accuracy, and coverage of data to be gathered) and (2) an adequate quality assurance plan against which achievement of data quality objectives can be reviewed. Not only are these steps essential in ensuring the cost-effective deployment of resources, their scope and design should closely support the goals and objectives of the overall study. Inadequacies in monitoring program design can seriously compromise the ability of a study to define the exposures and potential risks it seeks to investigate.

As the studies reviewed here illustrate, cost constraints inevitably lead to design compromises or trade-offs among multiple competing objectives. Any ambient air quality monitoring program is inherently expensive, but air toxics studies tend to be relatively more expensive than conventional programs. Sampling techniques are often not standardized, per-sample laboratory costs are comparatively high, and the number of compounds of potential interest is large. Furthermore, in the absence of specific ambient air quality standards for toxic air pollutants (which would define specific statistics of regulatory concern), study objectives tend to include multiple aspects of air toxics exposures: emission rates, fenceline concentrations, source-receptor relationships, average population exposures, or maximum individual exposures. It is therefore highly important for designers to reach firm agreement on a monitoring program's objectives, especially in terms of how these objectives relate to the broad technical questions of the study as a whole.

Examples of technical trade-offs that should be addressed in designing a monitoring plan include:

- Sample accuracy: The most accurate sampling and laboratory procedures are not necessarily the most desirable for every study. Achieving highly accurate results for individual samples may compromise the number of sites sampled, the number of compounds evaluated, the frequency of sampling, or the length of the sampling period. Undue emphasis on individual sample accuracy may actually reduce rather than enhance the effectiveness of the measured data in supporting program goals. This is especially true in cancer assessments because of the inherent uncertainty that exists in cancer unit risk factors.

- Site selection: Many factors influence site selection, including the gradient of concentrations expected in each area (locations with steep gradients require the most careful location of sites), laboratory detection limits (location of sites in areas where concentrations often fall below detection limits may waste resources), or air quality statistics of concern (average exposures versus maximum exposures).

The studies under review here illustrate these points in various ways. Some of the reports produced by these studies listed specific, formal monitoring objectives; for others, we have had to infer monitoring goals from the stated overall study objectives and technical approaches taken.

The advantage of presenting detailed, specific objectives prior to initiating a monitoring program is that it provides a focal point for consensus between program managers and technical analysts. In addition to helping these groups collaborate on a detailed and cost-effective QA/QC plan, setting formal objectives provides a benchmark to assess how well a program meets its goals.

Table 2-1 Summarizes the stated or implied monitoring goals of the eight studies with monitoring programs. Before examining individual technical issues in more detail, it is useful to discuss the stated goals of these eight monitoring programs and the extent to which they were achieved.

Studies with Stated Goals for Air Quality Monitoring Programs

The IEMP programs in Kanawha Valley, Philadelphia, Baltimore, and Denver; the Clark County Study; and the Urban Air Toxics Monitoring Program and the South Coast MATES project all documented their monitoring objectives.

Kanawha Valley—Designers of the Kanawha Valley program listed five general monitoring program objectives that clearly supported overall program goals for this unique industrial area:[1]

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[1] These are quoted verbatim from the study report.

Table 2-1. Summary of Objectives for Air Toxics Monitoring Programs Reviewed in This Study

	USES			SCALES ADDRESSED	
Study	Estimate Average Exposures	Evaluate Model Performance	Support Emissions Verification	Define Population Exposures	Define Maximum Concentrations
South Coast—MATES	X	X	X	X	X
Clark County	X			X	
IEMPs					
Philadelphia		X	X	X	X
Baltimore		X	X	X	X
Santa Clara	---------Goals Unspecified---------			------Goals Unspecified------	
Kanawha		X	X		X
Denver	X			X	
Urban Air Toxics Program	X			X	

1. To qualitatively and quantitatively confirm the presence of targeted compounds in the ambient air. EPA Region III's minimum objective was to confirm the presence of compounds for which emissions data were available from the West Virginia Air Pollution Control Commission's inventories. There was concern that some emissions data may have been outdated or otherwise inadequate and that modeled predictions of concentrations of certain pollutants could be in error, both because of technical difficulties in modeling the valley (see Chapter 4) and because of possible chemical reactions during transport.

   This goal was largely achieved by the program.[2] Most predicted compounds were documented, though ambiguities remained with regard to ethylene oxide.

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[2] Although the overall objectives of this study were met fairly well, Phase I of the two-phase monitoring plan developed for the study did not meet its study objectives. The strategy of Phase I was to conduct a limited three- or four-day screening study using bag sampling with GC analysis (and some GS/MS analysis for confirmation) to rank potential sites and select pollutants for the larger-scale effort in Phase II. Unfortunately, the GC and GC/MS results were not in agreement and no clear pattern emerged to distinguish upwind and downwind sites near major sources.

2. To determine whether a more in-depth emissions verification of facilities and processes is needed. To the extent that models could predict ambient concentrations, the study sought to determine which compounds in the inventories were least well characterized and required additional follow-up.

   This goal was also achieved. The study did find that certain measured concentrations did appear to be significantly higher or lower than predicted. As of this writing, however, the study has not reconciled specific differences by following up to determine whether certain emissions were overestimated or underestimated.

3. To assist in model selection for potential future study. Technical difficulties in modeling the Valley (see Chapter 4) are substantial. To fulfill this monitoring goal, the study data were to be interpreted to the extent feasible to determine whether model modifications should be attempted to better define transport and dispersion for this area.[3]

   This goal has not yet been achieved because the necessary analysis has not yet been done.

4. To help determine whether a more detailed monitoring program is needed at a later date. If significant most exposed individual MEI concentrations of study pollutants were detected in populated neighborhoods, and if monitoring confirmed that modeled results reasonably predicted the presence and locations of these MEI concentrations, additional monitoring might be appropriate to confirm model predictions in other locations or for additional compounds.

   This goal was achieved; predicted high concentrations of certain compounds were tentatively confirmed in target neighborhoods, indicating the appropriateness of further monitoring.

5. To evaluate a newly designed 24-hour canister sampling system. The Kanawha project used a relatively new evacuated canister system for sample collection. The final goal of the program was to evaluate the effectiveness of this method for the compounds of interest.

   The program confirmed the general effectiveness of the canister system.[4]

Overall, the Kanawha program clearly stated its monitoring program goals and documented its progress in meeting these goals in its final report. It was the only study reviewed that tracked achievement of its stated goals. Further discussions in this chapter and elsewhere will deal with how these program goals were reflected in details of the monitoring plan itself.

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[3] Specific issues included whether increased use of turbulent intensity data and a specific treatment for Valley wall reflection would be possible and desirable.

[4] Difficulties were encountered, however, with the formaldehyde sampling techniques used in the study, invalidating results for this pollutant.

Denver—The Denver IEMP project also listed general objectives for its monitoring effort that directly supported overall study goals. They were, however, markedly different in many respects than the Kanawha effort.

1. Collect ambient concentration data on selected toxic air pollutants that are likely to be present in metropolitan Denver using the best available sampling and analysis techniques. Denver's air pollution problem is considered to be regional: there are few industrial sources, and much of the air pollution is attributable to complex meteorological conditions and possibly transformation of pollutants from automobiles and other dispersed sources. Study designers determined that in this case it was necessary and appropriate to use the most current sampling methods available. The generalized nature of the pollution implied that concentrations would not vary extensively from site to site, permitting use of a small number of stations. The main issue of concern was to quantify multiple pollutant species in depth, using the most current methods available from EPA's Office of Research and Development.

   This objective is expected to be achieved. Data analysis is still in progress.

2. Collect ambient concentration data during high episodes for the routinely collected pollutants, and conduct broad-scan analyses[5] for those not routinely measured and which may warrant further study. This program objective called for increased 12-hour coverage of VOCs on days with peak CO and TSP concentrations.

   This objective was not fully achieved despite the use of independently analyzed 12-hour samples taken during predicted peak air pollution days. The goal of collecting additional samples on peak days was limited to one extra day of sampling.

3. Produce ambient measurements that are spatially and temporally representative of annual and acute [short-term[6]] exposures experienced by the general population. This program objective overlaps somewhat with objective 2. It called for distribution of sampling stations over the general

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[5] "Broad-scan" refers to a detailed GC/MS library search for each peak to attempt to estimate the concentration of as many pollutants as possible. More typically, analysis is limited to a preselected number of routinely quantified peaks.

[6] Use of the word "acute" implies short-term concentrations, not concentrations high enough to produce acute toxic responses in humans.

metropolitan area, sited both to reflect annual average concentrations and to capture high short-term concentrations.

Spatial representativeness was achieved, but, as noted, definition of short-term peak concentrations does not appear to be fully achieved.

4. Help identify and prioritize future research needs. The program's final goal was to set priorities for future studies in terms of compounds of interest and spatial and temporal resolution. It was expected that some compounds for which risk values are not available may be identified at relatively high levels, and that final evaluations of risk would be postponed until suitable potency estimates are forthcoming.

   It is not possible to assess the attainment of this goal until data interpretation is complete.

Philadelphia and Baltimore—These projects were among the earliest, if not the first, projects to attempt urban scale monitoring for air toxics. They had one basic monitoring goal—to use a source-oriented siting approach to help evaluate and verify local emission inventories. They were both partly successful in reaching this goal, though perhaps more elaborate and comprehensively developed goal statements could have improved the program results.

Clark County—The objective of this study was to determine annual average concentrations of most volatile organic compounds, selected metals, and asbestos. These compounds were expected to represent toxics of concern within the Las Vegas study area, which is heavily influenced by mobile sources. The study appears to have largely achieved its stated monitoring objectives.

Urban Air Toxics Monitoring Program—This is a continuing, multi-city program in which each monitoring study reflects the general objectives of the overall program. These objectives include the following:

1. To provide estimates of annual concentrations of selected air toxics.
2. To provide information for prioritizing and planning future work and sampling on a more in-depth and pollutant-specific basis in local areas.
3. To provide a means to identify prevailing pollutants and possible source types that may need further assessment.

4. To identify a means to evaluate and rank future air toxics mitigation programs.

South Coast—MATES—The South Coast Air Quality Management District (SCAQMD) developed a very detailed monitoring protocol to cover all important aspects of ambient sampling, as itemized below:

- Monitoring objectives
- Chemical species measured
- Sampling methods
- Analytical methods
- Identification of potential sites
- Sampling strategy
- Data quality objectives
- Quality assurance considerations

Specific monitoring objectives were:

1. Measure multiple toxic air contaminant concentrations in areas where the probability of the occurrence of elevated risk is greatest.
2. Collect data representative of the potentially toxic chemical species that are expected to be significant in the South Coast Air Basin.
3. Measure both organic compounds in the gaseous phase and metals on suspended particulate matter.
4. Collect samples that are representative of a winter/spring season and a summer/fall season, and from which meaningful estimates of annual average concentrations may be determined.
5. Collect data that can be compared with information from other ongoing toxics programs in the SCAB.

Details of the MATES monitoring program are given in (Shikiya, 1988).

2.3. Pollutant Coverage

Table 2-2 presents the pollutants covered in each monitoring program. As is clear from this table, studies to date have emphasized volatile organics and, to a considerably lesser extent, metals. Coverage of semivolatile organics and aldehydes/ketones has been very limited.

The focus on volatile organics was apparently a result of emphasis placed on volatile organics in available emission inventories, which often preceded these studies. Even where these inventories were developed expressly to evaluate toxic air pollution (as in Philadelphia), they tended not to look comprehensively at all categories of potential toxics, but to emphasize the high-volume, often carcinogenic, organic solvents. The implicit assumption was that these high-volume carcinogens were likely to account for the major fraction of the risk from toxic air pollutants. This bias toward volatile organics is clearly evident in the IEMP studies in Philadelphia, Kanawha Valley, and Santa Clara, whose goals were explicitly linked to verification or review of available emission inventories. These studies included coverage of pollutants released from industrial sources, as well as those related to the fueling, combustion, or evaporative emissions from mobile sources. Pollutants in this latter group include benzene, toluene, ethyl benzene, and the isomers of xylene.

Once the emphasis on volatile organics was established, most studies covered pollutants that could be measured by readily available methods. For the majority of studies conducted during the period of 1980 through 1985, Tenax®/GC or GC/MS methods[7] generally dictated pollutant coverage, producing a heavy emphasis on common solvents such as perchloroethylene and trichloroethylene (solvents such as methylene chloride and 1,1,1-trichloroethane tended to pose contamination problems). In addition,

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[7] The only other generally available method for organics sampling was to use charcoal as the sorbent, but charcoal was generally considered substantially inferior to Tenax.

Table 2-2. **Pollutants Covered In Ambient Air Monitoring Programs ***

POLLUTANT	Clark County Study	Urban Air toxics Monitoring Program	Staten Island Study	South Coast MATES	Santa Clara IEMP	Kanawha IEMP	Phil. IEMP	Denver IEMP	Baltimore IEMP
HALOGENATED ALIPHATICS									
Allyl chloride									
Bromochloromethane		X							
Bromodichloromethane		X							
Bromoform		X							
Carbon tetrachloride	X	X	X	X	X	X	X	X	X
Chloroethane		X							
Chloroform	X	X	X	X	X	X	X	X	X
Chloroproprene		X							
Chloropropene 3-								X	
Dibromochloromethane		X	X						
Dibromochloropropane									
Dibromodichloromethane									
Dibromoethane 1,2- (EDC)								X	
Dichlorobenzene 1,1									
Dichlorobenzene 1,3		X							
Dichlorobenzene 1,2									
Dichlorodifluoromethane								X	
Dichloroethane 1,1-		X						X	
Dichloroethane 1,2-		X	X	X		X	X	X	X
Dichloroethene 1,1-								X	
Dichloroethene cis-1,2-								X	
Dichloroethene trans		X							
Dichloropropane 1,2-		X					X	X	X
Dichloropropene cis-1,3-		X						X	
Dichloropropene trans-1,3-		X						X	
Dichloro-1,1,2,2-tetrafluoroethane 1,2-								X	
Epichlorohydrin									
Ethyl bromide									
Ethyl chloride						X		X	
Ethylene dibromide				X					

Table 2-2. Pollutants Covered In Ambient Air Monitoring Programs *

POLLUTANT	Clark County Study	Urban Air toxics Monitoring Program	Staten Island Study	South Coast MATES	Santa Clara IEMP	Kanawha IEMP	Phil. IEMP	Denver IEMP	Baltimore IEMP
Ethylene chloride									
Hexachlorobutadiene								X	
Methyl bromide		X						X	
Methyl chloride		X	X					X	
Methylene chloride		X	X	X	X	X	X	X	X
Perchloroethylene	X	X	X	X	X	X	X	X	
Tetrachloroethane 1,1,1,2-									
Tetrachloroethane 1,1,2,2-		X						X	
Trichloroethane 1,1,1-		X	X	X		X		X	
Trichloroethane 1,1,2-		X						X	
Trichloroethylene		X	X	X	X	X	X	X	X
Trichlorofluoromethane								X	
Trichloro-1,1,2-trifluoroethane 1,1,2-								X	
Vinyl chloride	X	X	X	X		X		X	
INORGANICS AND METALS									
Asbestos	X								
Arsenic	X	X	X	X				X	
Barium		X	X	X				X	
Beryllium		X	X	X				X	
Cadmium	X	X	X	X				X	
Chromium (total)	X		X	X				X	
Cobalt		X	X	X				X	
Copper		X	X	X				X	
Iron		X	X	X				X	
Lead		X	X	X				X	
Manganese		X	X	X				X	
Mercury									
Molybdenum		X	X					X	
Nickel	X	X	X	X				X	
Titanium dioxide									
Vanadium				X					
Zinc		X	X	X				X	

Table 2-2. Pollutants Covered In Ambient Air Monitoring Programs •

POLLUTANT	Clark County Study	Urban Air toxics Monitoring Program	Staten Island Study	South Coast MATES	Santa Clara IEMP	Kanawha IEMP	Phil. IEMP	Denver IEMP	Baltimore IEMP
MONOCYCLIC AROMATICS									
Benzaldehyde								X	
Benzene	X	X	X	X	X	X	X	X	X
Benzyl chioride								X	
Chlorobenzene		X				X		X	
Dichlorobenzene m-								X	
Dichlorobenzene o-								X	
Dichlorobenzene p-								X	
Ethyl benzene							X	X	X
Ethyl toluene 4-								X	
Methyl Thiozole			X						
Nitrobenzene									
Styrene		X						X	
Terephthalic acid									
Tolualdehyde m-								X	
Tolualdehyde o-								X	
Tolualdehyde p-								X	
Toluene		X	X	X	X	X	X	X	X
Trichlorobenzene 1,2,4-								X	
Trimethyl benzene 1,2,4-								X	
Trimethyl benzene 1,3,5-								X	
Xylene m-		X			X		X	X	X
Xylene o-		X	X		X	X	X	X	X
Xylene p-		X			X		X	X	X
NON-HALOGENATED ALIPHATICS									
Acetaldehyde		X	X					X	
Acetone								X	
Acrolein		X	X					X	
Butadiene 1,3-		X							
Butyraldehyde								X	
Decane n-									
Dioxane 1,4-									
Dodecane									
Ethyl Acrylate									
Ethylene	X								

Table 2-2. **Pollutants Covered In Ambient Air Monitoring Programs ***

POLLUTANT	Clark County Study	Urban Air toxics Monitoring Program	Staten Island Study	South Coast MATES	Santa Clara IEMP	Kanawha IEMP	Phil. IEMP	Denver IEMP	Baltimore IEMP
NON-HALOGENATED ALIPHATICS (cont'd)									
Ethylene oxide			X						
Formaldehyde	X	X	X	X				X	
Hexane			X						
Pinene alpha-									
Propionaldehyde								X	
Propylene oxide									
Butyl cellusolves									
Methyl cellusolves									
POLYCYCLIC AROMATICS									
Benzo(a)pyrene		X	X	X				X	
Benzo(g,h,i)perylene								X	
Benz(a)anthracene								X	
Chrysene								X	
Coronene								X	
Cyclopenta(c,d)pyrene								X	
Fluoranthene								X	
Isoquinoline								X	
Melamine									
Methylpyrene								X	
Naphthalene								X	
Nitropyrene								X	
Phenanthrene								X	
Pyrene								X	
Pyrene								X	
Quinoline								X	

* See Chapter 1 for a description of each program.

Tenax® is not suitable for measuring pollutants such as vinyl chloride, 1,3-butadiene, or acrylonitrile, which is why these organics were not included in early studies.

Trends Toward Broader Pollutant Coverage

Recent studies are moving toward more comprehensive pollutant coverage. Volatile organics are still an important class of pollutants and are given major emphasis, but there is a growing realization that metals, semivolatile compounds, aldehydes/ketones, and possibly asbestos may contribute significantly to potential health risk in certain urban locations. This, in conjunction with improvements in sampling methods,[8] supports a freer and more inclusive selection from multiple classes of toxic pollutants.

For example, the South Coast MATES selected pollutants for inclusion in the monitoring programs based on four factors: toxicity, emissions inventory coverage, comparability with ongoing toxics programs, and constraints of sampling and analytical methods. This systematic approach led to the coverage of 6 metal and 14 volatile organic compounds, with a fairly balanced mix between these two major classes. The Clark County Study also chose a mix of VOCs and metals, and included some asbestos monitoring as well.

Similarly, the more recent Staten Island and Denver studies were designed for relatively comprehensive pollutant coverage, with volatile organics, metals, and aldehydes included. Denver also considered semivolatile organics.[9] The more recent studies broaden our understanding of the magnitude and variability of the components of the urban soup. The data collected by the Staten Island and Denver monitoring programs, in particular, have greatly expanded the coverage into each of the major classes listed above.

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[8] For example, although Tenax is generally not well suited for sampling methylene chloride and vinyl chloride, these and other compounds can now be sampled by canister methods, which are replacing Tenax for many applications, especially for fixed-station use.

[9] For reasons discussed under site selection, the Denver project also monitored for CO and fine particulates to help extrapolate data for broader spatial coverage.

2.4. Site Selection

A variety of site selection methods were used in these studies. Differences in approaches generally relate to differences in the objectives of each study, but study objectives must inevitably be balanced against practical siting considerations. Where studies have attempted to support multiple monitoring objectives, and most have, siting has involved a complex series of tradeoffs.

Of the studies reviewed, four used monitoring expressly to estimate average population exposures. Some sited monitoring stations according to explicit technical criteria (examples include the South Coast MATES, Philadelphia, Baltimore, Kanawha, and Denver studies); others approached the problem using no prescribed protocol (the Staten Island and Santa Clara studies).

Formal [Quantitative] Approaches to Site Selection

South Coast MATES—The objectives of the ambient monitoring portion of this study were to quantify maximum individual risk, estimate average background contaminations, and assess model performance and its emission inputs. Eleven existing SCAQMD sampling sites were used to estimate average background concentrations and to compare with SCREAM model outputs. Ten new sampling sites, selected by a method outlined below, were meant to be located in areas of high concentrations, and thus indicate maximum individual risks.

The study developed a three-step approach to site selection that could be adapted to meet needs in similar locations. Site assessment analysis covered a large area—70 by 60 kilometers—or 4,200 1 km^2 grid cells, based on the 1 km grid spacing used in the study's dispersion modeling and for site selection.

In step one, SCAQMD conducted dispersion modeling of emissions to estimate annual average concentrations of each individual target compound within each grid cell. These concentrations were then normalized as ratios between each predicted concentration and the 40th highest concentration for the pollutant in question. (This ratio provides an indication of the strength of the concentration gradients for each pollutant.) The ratios in each cell were then summed across pollutants. According to the study report, this procedure identified those areas with the greatest exposure potential for site ranking purposes.

In step two, the designers selected ten separate clusters of cells as candidates for monitoring sites, based on the following considerations:

1. Emphasizing cells with maximum concentration estimates above or close to detection limits for the test pollutants;
2. Giving priority to cells with the highest concentration estimates;
3. Providing adequate coverage of mobile and stationary sources; and
4. Emphasizing cells with relatively high overall potential health risk, considering the unit risk associated with individual pollutant species.

The first two considerations essentially sorted out cells with the highest individual concentrations and the highest sums of ratios. Using the third and fourth considerations, the study selected final monitoring clusters based on representativeness of multiple source categories, as well as high potential health risks.[10]

In the third step of the site selection process, the study considered practical siting factors, such as availability of power, security, access, source proximity, receptor profile, species mix, and micrometeorological factors.

Details of SCAQMD's site selection protocol are in (Shikiya, 1988).

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[10] The criteria for balancing these factors, whether qualitative or quantitative, are not discussed in the study documentation.

Denver—The IEMP Denver study used a rather different, but also formally structured, approach to monitoring station siting. The site selection analysis used is based on some unique factors pertinent to this metropolitan area, such as mobile source dominance, but may have applicability to other sites where characterizing broad regional variations in the urban soup is the goal, and industrial sources do not create substantial gradients in concentrations. This study's site selection problems were simpler, since model performance evaluation and emissions inventory verification were not among the study's objectives.

The study objectives called for locating three sites at which detailed, in-depth, broad-spectrum sampling and analysis would be conducted. Because data on emissions and ambient concentrations of air toxics were limited at the outset of this study, the designers used criteria pollutant data as surrogates for selected classes of toxic air pollutants in the site selection process.

The site selection process involved four steps. In step one, available criteria monitoring data from various monitoring stations were analyzed statistically. The study computed means and standard deviations of pollutants at each monitoring location and conducted paired t-tests to compare means among sites.

In step two, sites were grouped into neighborhoods that showed the closest correlations and summary statistics. Defining neighborhood groups involved some judgment, however. For instance, CO data were given more weight than the other criteria pollutants because previous studies had concluded that mobile sources would be the most significant toxics contributors.

Step three selected three neighborhoods that study designers believed best represented the region as a whole. Again, although the supporting analysis was quantitative, some judgment was involved.

In step four, actual sites were located within the three selected neighborhoods, based mainly on practical operational considerations. At this time, however, one of the three sites was located nearer to a traffic corridor—a deviation from the general program objective of establishing average annual concentrations—to better define the high range of air toxics concentrations from mobile sources. To compensate for any resulting bias in regional exposure estimates, the study established a fourth monitoring site to collect CO and fine particulate data as an aid in interpreting the results of the monitoring program. Although the selection of this fourth site resulted in a departure from formal study objectives, the effort to quantify possible biases is a practical example of compromises that are often needed to achieve multiple objectives in an air toxics monitoring program.

Philadelphia and Baltimore—If dispersion modeling is used to estimate risks, the comparison of measured and modeled data can reveal how well the model is performing and point to where model performance can be improved. Criteria for locating sites in these two studies were similar to those discussed in the South Coast study. Again, the principal concern was to ensure that the maximum number of samples show concentrations above laboratory detection limits, and that there is good representation of source categories contributing detectable concentrations at each monitoring site.

The Philadelphia and Baltimore monitoring programs also emphasized emission inventory verification.[11] The siting strategy in these two projects, therefore, emphasized siting in industrial neighborhoods where concentration gradients of targeted VOC pollutants were expected to be steepest.

The IEMP Philadelphia and (to a lesser extent) Baltimore studies used dispersion modeling in selecting neighborhoods for coverage in the monitoring program. Neighborhoods (e.g., 1 km by 1 km areas) expected to have the sharpest gradients in ambient concentrations, as well as the highest concentrations, were identified through the use of screening-level modeling.[12] Network coverage was sought that provided representative coverage within each selected neighborhood. Actual selection of sites was limited by the finite number of practical sites within many of the neighborhoods identified for inclusion in the monitoring program. The primary problem was identifying sites that were secure, accessible, and not unduly affected by minor local sources, such as adjacent roadways, gas stations, and so forth.

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[11] The South Coast study also was concerned with evaluating model performance, but was equally concerned with estimating average annual exposures—not a formal objective in Philadelphia and Baltimore. This in part explains the somewhat different siting strategy used by the South Coast study.

[12] This is similar to the analytical approach used in the South Coast study, but the IEMP modeling analyses were much less detailed than in the South Coast study.

The Philadelphia study provides a special case objective for testing model performance near two major sources. In the Philadelphia study, two of the ten sites were selected to be in the vicinity of major point sources to help assess maximum concentrations in these areas. The remaining sites were selected to be more representative of typical exposures in residential neighborhoods. For the two special-case monitoring sites, the goal was to be as representative as possible of maximum concentrations.

For the two special sites, the selection process was guided by review of the emissions inventory rather than by dispersion modeling. One area of concern was near a major pharmaceutical company, located downtown. The actual monitoring site was selected approximately 500 m from the source, within the same street canyon. This distance was chosen because of concerns that vent releases from the large building might not mix effectively prior to reaching a closer monitoring site. Unfortunately, based on records collected from the facility, emissions of chloroform, the key pollutant of concern at this source, were highly variable. Significant emissions were measured only during a few days of the 30-day monitoring program, and at that time there was little wind flow toward this site. The objective of evaluating impacts near this source was therefore not effectively achieved.

The second special monitoring site was placed near the major refineries in the southeast section of Philadelphia. The station was established just downwind of this major industrial complex, approximately 500 m from the fenceline. Because emissions from this source region were more uniform and because the wind flow toward the monitoring site was more frequent, this site achieved its objective. Monitoring data from this station helped to corroborate emissions data, indicating lower emissions rates than anticipated at the outset of the study.

Kanawha Valley—Site selection in this study area was simplified because the emissions of toxic air pollutants were heavily clustered into three separate areas located within the confines of a narrow, 1 to 2 km wide, river valley. Two of the three areas were selected based on the magnitude of emissions of toxic pollutants and the amenability of the canister sampling method to cover the key pollutants emitted by the industries in these areas, i.e., the Belle Complex (which included a Diamond Shamrock and DuPont plant) and the Institute Complex (which included a Union Carbide plant).

Kanawha Valley is a good example of how the availability of monitoring methods can heavily influence site selection. The emissions data that were available during the design stage of the monitoring program suggested that the Institute and South Charleston complexes were of greatest priority in terms of risk—a tentative hypothesis subsequently borne out by more detailed emissions modeling data. Belle was selected for a monitoring site in place of South Charleston because the solvents released in Belle (e.g., carbon tetrachloride, chloroform, methylene chloride) were more amenable to canister sampling than the pollutants of greatest concern in the South Charleston complex, namely ethylene oxide and acrylonitrile.

The narrow valley width facilitated the selection of an upvalley and downvalley site in each of these two areas, because the major consideration was distance from the boundaries of these tightly clustered industrial areas. With few exceptions, the major releases of toxic air pollutants in the Kanawha Valley are from near-ground-level sources, which would be expected to produce maximum ambient annual average impacts near or at the fenceline of the plant boundaries. This factor together with the restricted flow produced by the terrain greatly simplified the search for sites by making it essentially a one-dimensional problem.

For this study, sites were located within a specific range of between 0.5 km and 2 km from industrial fencelines, rather than at residences directly adjacent to the fencelines (i.e., within 100 m of the property boundaries). This was done because the emphasis of the monitoring study in Kanawha Valley was on evaluating the reasonableness of local emissions inventory data. The 0.5 km to 2.0 km distance range was selected because, with only 15 days of sampling to be available, the study's designers were concerned that plumes from industrial complexes might travel around or beside monitoring sites located closer than 0.5 km to the fenceline. There was also the possibility that plumes might extend above the monitors on specific days. Both eventualities would complicate interpretation of such a comparatively short (15-day) data set. The 2 km maximum distance was selected because the Gaussian models used in the study would produce more uncertain results for greater distances because of the influence of the valley walls and complication in flow behavior. The measured and modeled data were in reasonable agreement, suggesting that the modeled data were at least within the right order of magnitude and thus were achieving one of the basic objectives of the study.

Informal [Qualitative] Approaches to Site Selection

Staten Island—The Staten Island study is an example in which selection of monitoring sites does not rely on analysis of available measured or emissions data. Instead, this study sought to estimate exposures mainly by establishing wide coverage throughout the geographic bounds of the study area—a more subjective approach.

While more subjective placement of samplers was generally performed in the Staten Island study than was performed in South Coast, established guidelines were followed for site selection, especially at the microscale. EPA guidance for SO_2 was followed in locating volatile organics sampling sites; EPA's particulate guidance was used for the high volume samplers collecting metals data. These guidance procedures are focused principally on microscale siting factors such as the height for sampling, distance from nearby sources, and so forth, rather than on selecting neighborhoods analytically in relation to sources and predicted concentrations.

The main limitation of this more subjective approach to siting monitors involves the interpretation of the relationship between average concentrations at the monitoring sites and the average concentrations in neighborhoods elsewhere throughout the study area; such relationships need to be established in order to estimate exposure based on measured data. While it is possible to complete this step after data collection, the main risk is that some of the sites may be found to be in areas that do not best represent exposures within the study area. At that point, the data would need to be extrapolated to better estimate more general exposures. On the other hand, if this were found during the site selection stage, a different, more representative site could be selected. This is in contrast to South Coast MATES where the representativeness of monitoring sites to the full receptor array could be more quantitatively demonstrated based on the modeling results prior to the collection of the first sample.

Urban Air Toxics Monitoring Program—In this study, single sampling sites were informally selected to be broadly representative of urban exposures and not influenced strongly by nearby point sources. (Note: multiple sampling sites were encouraged, but most participating agencies had inadequate resources for more than one site.)

Santa Clara—The Santa Clara study provides another example of qualitative site selection. In this study, site selection was limited to existing air quality monitoring sites. Five existing air quality monitoring sites, part of the Bay Area Air Quality Control Region's existing criteria pollutant network, were selected. One of the five sites was in Gilroy, a town near the southern extreme of the study area, in order to provide background data[13] to compare with the more heavily developed northern portion of the study area. The remaining four sites were selected to obtain coverage in the more urban commercial and residential areas. None of the existing sites was adjacent to a major industrial facility; instead, they were chosen to be more representative of selected neighborhoods, distributed as widely as possible throughout the study area. In this sense, site selection goals were similar to the Staten Island Study.

The Staten Island and Santa Clara studies have used similar site selection procedures, but it appears that the Staten Island study is more likely to benefit from the measured data set. The primary reason is the large spatial and temporal coverage of the

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[13] The concept of a background site has benefits in terms of interpreting the measured data set, and potentially for assessing background values for monitoring analysis. Because most studies can afford only a limited number of sites, background stations generally have not been established—none except this one was noted in any of the studies reviewed.

Staten Island study. The 13 site network, collecting data over nearly two years, is developing a large reservoir of data to partition and interpret as a function of wind flow and other factors. This richness provides information to help separate potentially unrepresentative data from data that better represent typical concentrations. Furthermore, sites found to be generally unrepresentative on this basis could be dropped from future exposure assessments, if necessary. The Santa Clara project, on the other hand, with 5 sites collecting data over just 5 days, did not provide enough data to support an exposure assessment or model performance evaluation. Moreover, the limited quantitative understanding of the selected sites further weakened the utility of this data set.

Physical Siting Criteria Common to All Studies

Certain common practical considerations affect any ambient air monitoring program. Studies of toxic air pollution are no exception, and in fact may often be more sensitive to certain localized factors than studies of most criteria pollutants. In selecting sites for an air toxics ambient air monitoring network, designers must be sure that each site meets the following general criteria:

- Representativeness—The goal typically should be to collect data that would be representative of a broad area, such as a neighborhood. In general, sites should be selected to avoid local sources and flow interference from nearby structures or vegetation, including micrometeorological influences from nearby hills, bodies of water, valley drainage patterns, and so forth. Sampling line intakes should be within a typical range of 2 to 10 m in height above the ground. The probe should extend at least 2 m from a supporting structure; if located on a building, it must be mounted on the windward side. The distance between any obstruction and the sampler should not be closer than two times the height of the obstruction.

- Physical security—Securing instruments from tampering or theft often has led to rooftop sampling or the use of existing air quality monitoring sites that are part of local criteria pollutant monitoring networks. When new sites must be selected, rooftop monitoring is frequently used. For these sites, the main issue was the avoidance of localized emissions from roofing material or vent releases. The use of rooftop sampling for urban soup and other studies of toxic air pollutants have been questioned, however. (Wallace, 1987) Documentation was not available for any of these studies to demonstrate that rooftop sampling did not interfere with the collection of representative data.

- Adequate power and access—These are operational factors concerning issues such as the availability of power for all instruments and support equipment, space considerations, 24-hour access, etc., that enter into the site selection process.

The above factors impose practical constraints that are common to air toxics or criteria pollutant monitoring programs and will not be addressed further in this report. To the best of the authors' knowledge, they were followed in all of the studies reviewed here.

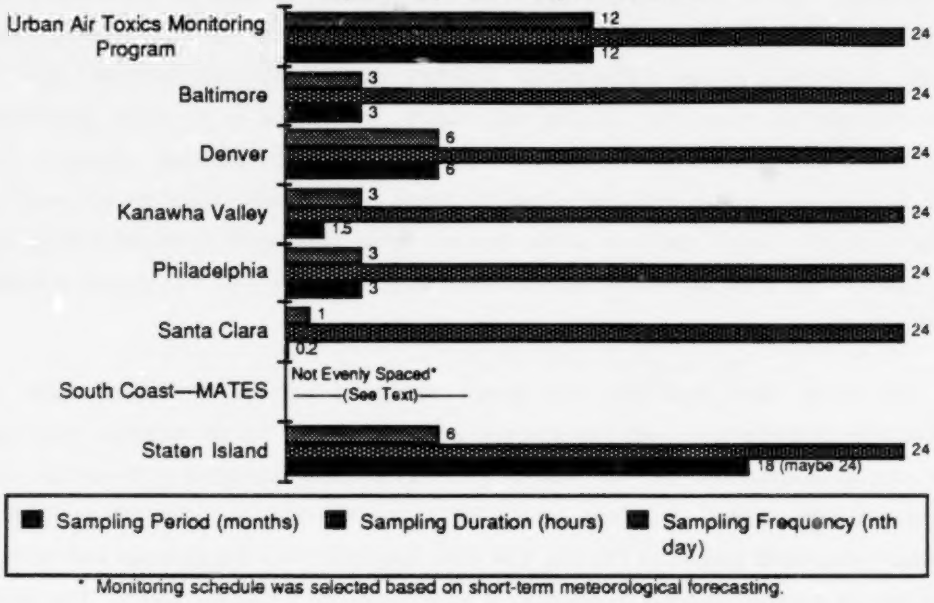
2.5 Sampling Periods, Frequency and Duration

The primary averaging period for these studies was annual because annual average concentrations were used as the basis for estimating exposures to carcinogenic air pollutants. Increasing the number of hours that monitors operate (assuming an unbiased sampling schedule) should improve the accuracy with which measured data can estimate annual average concentrations. Three interrelated design factors determine the number of hours used to estimate seasonal or annual averages at each site in a sampling network: (1) the sampling period, (2) the sampling duration, and (3) the sampling frequency.

Sampling period refers to the length of time during which the field program is operational, from the first sampling day to the last. The number of hours that compose each sample is the sampling duration. Sample frequency is the manner in which days for sampling are separated, e.g., every second day, every third day, every sixth day, and so forth.

Figure 2-1 presents the sampling periods, sampling durations, and sampling frequencies for each of the studies evaluated. As shown in Figure 2-1, there is a wide range in the extent of data collection across these studies, ranging from 5 days of coverage in Santa Clara, to 1.5 years of every 6th day sampling in the Staten Island project. In all cases, funding has been the key element affecting temporal coverage. Ideally, all studies that emphasize cancer risks, as reviewed here, would prefer complete monitoring over at least a one-year period. Since the cost of such monitoring is prohibitively expensive, program designers must generally define the minimum data needed to adequately meet each project's stated objectives.

Figure 2-1. Summary of Monitoring Periods, Sampling Frequencies, and Sample Duration



* Monitoring schedule was selected based on short-term meteorological forecasting.

NOTE: the Denver study also conducted limited 12-hour sampling.

Sampling Period

The selection of an appropriate sampling period is driven by the end use of the data set. For studies to date that evaluated cancer risks, there were two major approaches used: (1) use of measured data to directly estimate annual average concentrations, and (2) use of measured data to assess, and improve if possible, model performance. The second approach was then followed by predicting concentrations, and thereby exposure and risks, through the same basic modeling approach.

Denver—The IEMP Denver study used monitoring in the winter and summer seasons to estimate annual average concentrations for all pollutants. The study concluded, based on interpreting criteria pollutant data, that averaging data from the summer and winter periods provided reasonable estimates of annual average concentrations. These sampling periods were also selected to track periods with peak concentrations. Photochemically formed compounds peak during the summer and have minimum concentrations during the winter. The less reactive pollutants, such as benzene, carbon tetrachloride, and other volatile compounds, were expected to reach maximum concentrations during the winter and minimum values during the summer. Although the periods with generally high and low values for these different pollutants are opposite to one another in the annual cycle, in either case averaging the results collected during the summer and winter seasons was thought to reasonably represent peak and annual average concentrations.

To check this assertion, the study analyzed criteria pollutant data to quantitatively evaluate how well the summer and winter seasons on balance represent annual average concentrations. CO was used as the primary indicator pollutant for the dominant mobile source emissions, and TSP as an alternative indicator pollutant. Preliminary statistical review of CO and TSP data suggested that the summer and winter seasons would provide balance in covering a wide range of pollutant classes. For these criteria pollutants, estimates of annual average concentrations based on the summer and winter periods were within 10 to 15 percent of the annual averages as calculated using a full year of data.

South Coast MATES—At the 11 existing sites, sampling has been ongoing for an indefinite period. At the 10 new sites selected to capture high exposures, two sampling periods were defined, each five months long. One period corresponded to the winter/spring seasons and the second, the summer/fall seasons. Thus, samples were taken at the new sites during periods representative of an entire year.

IEMP in Philadelphia, Baltimore, Kanawha Valley—The IEMP studies in Philadelphia, Baltimore, and Kanawha Valley selected a sampling period of one season to serve as a checkpoint on model performance. For this objective, a single season appears to be a reasonable minimum sampling period so long as there are not likely to be strong, poorly characterized seasonal variabilities in the emissions of key compounds. For many of the volatile organics addressed, this is probably a reasonable assumption. Wood burning, however, is an example where distinct seasonal trends occur. The IEMP monitoring programs were performed during the winter and spring seasons primarily because of project schedule constraints.

Santa Clara—For this study, a five-day monitoring program was undertaken on consecutive days to obtain data to directly assess the general magnitude of concentrations at selected locations within the study area. The intent was to obtain some direct measurements to complement the modeling-based exposure assessment. In retrospect, it appears that the short duration of this program hampered the validity of comparisons with modeled data: the data set was too limited and short to reasonably assess general concentrations.

Sampling Frequency

Most of the studies used a sampling frequency of every 3rd, 6th, or 12th day. In general, the use of longer sampling frequencies and predetermined sampling intervals allows for: (1) extending the collection of measured data throughout a season, (2) avoiding bias introduced by a non-random selection criteria, and (3) obtaining more independent data than would be collected with monitoring during consecutive days.

The South Coast MATES study chose an alternative approach. Two ambient sampling efforts were conducted: (1) sampling at regular intervals at a network of existing sampling sites and (2) sampling at new sites believed to represent locations of maximum individual risk. Sampling at the 11 existing sites was conducted at regular intervals of about once every two weeks for organics and about once every week for metals. At two contributing EPA sites, sampling was conducted once every 10 to 12 days for benzo(a)pyrene and metals. At the 10 new sites, sampling days were selected based on short-term meteorological forecasting rather than on a regular sampling frequency. No samples were collected at the new sites on Saturdays or Fridays, since they could not be processed within 24 hours. (Barcikowski, 1988)

The approach used in the South Coast MATES project for the sites selected to represent locations of maximum individual risk is useful in that sampling days could be selected that best met project objectives (such as days in which wind flow is consistent or days with no precipitation). It offered the potential to fill a matrix of pre-selected conditions, which, if properly weighted, could also be used to represent annual average conditions. This approach, however, requires that the analyst show that the days selected for sampling adequately represent annual average conditions. In contrast, when using every 3rd, 6th, or 12th days regular sampling frequencies the random nature of the selection process should act to statistically represent the monitoring period without the need to demonstrate representativeness. Even the random selection of sampling days, however, could require a demonstration of representativeness to annual average conditions if only a small number of sampling days are included.

Sampling Duration

Concentrations can vary markedly as a function of time at any given site because of periodic variations related to diurnal cycles in meteorological conditions or emissions patterns. Industrial processes with rather uniform, low-level emissions may, for instance, show substantially higher concentrations caused by relatively poor dispersion at night. Mobile source emissions tend to be greatest during the day, while wood smoke and other residential-related emissions are generally higher at night. All of these cases point to the importance of considering diurnal variations when using measured or modeled data to support an exposure assessment.

Two basic ways that study managers have selected sample durations to account for this variability are as follows: (1) collect 24-hour samples to expand the number of hours integrated into each sample (and avoid potential bias by using less than 24-hour durations) or (2) collect sets of 12-hour daytime and nighttime samples to show the diurnal differences while retaining the ability to compute daily averages.

In the South Coast MATES project, all of the samples at the new sites were collected over a 24-hour period beginning at approximately 0900 Pacific Standard Time. The choice of starting hour was chosen so that data would be comparable to that from the "background" ambient toxics program, which was running concurrently. The rationale for choosing this particular 24 hour period as opposed to a midnight-to-midnight period was to collect samples over a contiguous meteorological drainage period (stable conditions) rather than splitting the drainage period. (Shikiya, 1988)

Of the operational studies reviewed, the Denver study was the only one that routinely collected 12-hour samples at some sites.[14] The other studies used 24-hour duration samples. The decision to use 12-hour samples for a subset of the program was made because of the following factors:

- The diurnal patterns in the Denver metropolitan areas are accentuated, especially during the winter season, by topographic complexities. Collecting 12-hour samples provided the opportunity to better interpret the measured data sets to show the relative importance of source categories and to better characterize peak concentrations.
- Receptor modeling was a secondary objective: this could be substantially enhanced by the collection of 12-hour samples because source culpabilities[15] could be more easily distinguished.

The advantage of collecting 12-hour diurnal samples is that incremental impacts from source categories can be better distinguished. The obvious disadvantage is that two samples, instead of one, need to be analyzed to represent one day, cutting in half the number of hours measured per monitoring dollar spent. Twelve hour samples are typically taken from 7 a.m. to 7 p.m. and from 7 p.m. to 7 a.m. each day.

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[14] Refer to Section 2.7 for a description of the TEAM and IACP, where 12-hour sampling also was done to help display diurnal differences in emissions, and to account for distinct differences in subject activity patterns.

[15] The "culpability" of a source refers to that proportion of a measured or modeled concentration at a particular geographic location attributable to that source.

2.6 Sampling and Analytical Techniques

Tables 2-3 and 2-4 present a summary of techniques used to sample and analyze, respectively, the classes of compounds measured in these studies. The wide range of methods can be briefly summarized by pollutant class: those pertaining respectively to volatile organics, semivolatile organics, metals, and aldehydes/ketones.

Volatile Organics

This class is historically the most widely covered. For the studies reviewed, there were two major methods used to sample VOCs: Tenax® and canisters, with Tenax® being absorbent material and the canisters collecting samples of ambient air. Generally, a combination of gas chromotography (GC) and gas chromotography/mass spectrometry (GC/MS) has been used in conjunction with these methods to analyze the samples.

The trend has been toward greater reliance on canisters because of the documented unreliability of Tenax®.[16] Even though canister sampling generally suffers from higher detection limits than Tenax®, canister sampling appears to offer greater day-to-day consistency in results and is therefore preferred for most applications,[17] despite its higher detection limits.

Uncertainties in using Tenax® can be reduced by using a distributed volume sampling protocol, in which four sampling tubes collect samples at different flow rates. (Walling, 1984) There is no evidence in the documentation of any of the studies reviewed here, however, to show that the distributed volume technique has been used when Tenax® was the sampling medium. Disadvantages of distributed volume sampling include a near quadrupling of analytical costs and relatively high rejection rates among samples tested. Adherence to this method effectively renders Tenax® unacceptable for operational studies.

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[16] See Walling 1984. Problems include (1) the apparent variability in effectiveness as a function of ambient temperature, relative humidity, and specific mixtures of pollutants, (2) vulnerability to breakthrough, and (3) vulnerability to contamination.

[17] An exception is for personal monitoring, where Tenax continues to be used (see Section 2.5).

Table 2-3. Summary of Sampling Techniques Used For Ambient Air Quality Monitoring Programs

SAMPLING METHODS	Clark County Study	Urban Air Toxics Monitoring Program	Six Month Study (Ambient Air Quality Component)	Staten Island Study	South Coast Study	Santa Clara IEMP	Kanawha IEMP	Philadelphia IEMP	Denver IEMP	Baltimore IEMP
Summa Cannister (volatile organics)	X	X	X	X			X		X	
Tenax (volatile organics)			X	X		X		X		X
Tedlar (polyvinyl fluoride) bags (gaseous compounds)			X		X					
Polyurethane foam (semi-volatile organics)									X	
DNPH cartridges (aldehydes/ketones)		X	X	X				X	X	
Dichotomous sampling (metals & carbon)	X		X						X	
X-ray fluorescence (metals)									X	
nitric acid/nitrate sampler (nitric acid/nitrate)									X	
Denuder-difference sampler (nitric acid/nitrate)									X	
Hi-volume sampler (metals)	X	X	X	X	X				X	
Hi-vol with Anderson impactor (fine particle for bioassay)										
Hi-vol with Sierra impactor (carbon 12/14)										
XAD-2										

Table 2-4. Summary Of Analytical Techniques Used For Ambient Air Monitoring Programs

TECHNIQUES	Clark County Study	Urban Air Toxics Monitoring Program	Six Month Study (Ambient Air Quality Component)	Staten Island Study	South Coast Study	Santa Clara IEMP	Kanawha IEMP	Philadelphia IEMP	Denver IEMP	Baltimore IEMP
Gas chromotography flame ionization detector (for VOC's)		X	X		X		X		X	
Gas chromotography electron capture detector (for VOC's)		X	X		X		X		X	
Gas chromotography photoionization detector (for VOC's - aromatics)		X	X		X			X		X
High performance liquid chromatography ultraviolet detector (aldehydes/ketones)		X							X	
Liquid chromatography fluorescence detector (for semi-volatile and particle-associated compounds)		X	X							
Gas chromotography/mass spectrometry (for volatile and semi-volatile, organics and particle-associated compounds)	X		X			X	X	X	X	X
Neutron activation analysis (metals)		X							X	
Atomic absorption spectrophotometry (metals)			X		X					
Inductively Coupled Plasma (ICP)		X								
Pyrolisis (carbon analysis)									X	

Table 2-4. Summary Of Analytical Techniques Used For Ambient Air Monitoring Programs

TECHNIQUES	Clark County Study	Urban Air Toxics Monitoring Program	Six Month Study (Ambient Air Quality Component)	Staten Island Study	South Coast Study	Santa Clara IEMP	Kanawha IEMP	Philadelphia IEMP	Denver IEMP	Baltimore IEMP
Gas chromotography/Hall Detector (chlorinated VOC's)								X		X
X-ray fluorescence spectrometry (metals)	X								X	
Polarized light microscopy (asbestos)	x									

Semivolatile Organics

The class with the most limited coverage among the studies reviewed is semivolatile organics. These samples can be collected on polyurethane foam (PUF) or on XAD-2, another sorbent compound. Analysis is by gas chromatograph/mass spectrometer (GC/MS), or by GC/MS coupled with high performance liquid chromatography (HPLC). XAD-2 is generally preferred when biological testing is to be performed, as it has been in the Integrated Air Cancer Project (IACP). The IEMP Denver study used PUF sampling and GC/MS analysis to cover this class of pollutants. One reason that the sampling of semivolatile organics is so limited is cost: analyses can range from $1,000 to $2,000 per sample, as shown in the documentation for the Denver study. (Machlin, 1986) The trend toward sample compositing for semivolatiles is an attempt to extend the number of hours integrated into each sample to stretch analytical dollars within the generally tight budgets of applied programs.[18]

Metals

Metals have been included in many urban studies. In the South Coast MATES and Urban Air Toxics Monitoring Studies, high volume samplers were used to collect metals samples, a different procedure from that used by the Denver studies, which collected fine particulate mass using fine particulate samplers. Atomic adsorption spectrometry, neutron activation analysis, inductively coupled plasma spectroscopy, and X-ray fluorescence spectroscopy are the methods generally preferred for metals analysis.

Hexavalent chromium (Cn^{+6}), an important contribution to urban air toxics risk, has not been distinguished from total chromium in most ambient measurements. Cr^{+6} was reported in ambient samples collected as part of the SCAQMD MATES project. (Shikiya, 1988) Many studies that have analyzed Cr^{+6} risk have either (1) assumed a certain percentage of total ambient chromium is Cr^{+6} (e.g., 10 percent or 100 percent) or (2) modeled Cr^{+6} and Cr^{+3} emissions distinctly.

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[18] See Section 2.8 for a more detailed discussion of sample compositing.

Aldehydes/Ketones

Formaldehyde and the remaining aldehydes/ketones were addressed by 2,4-dinitrophenylhydrazine (DNPH) cartridges and HPLC analysis in Denver, Staten Island, and the Kanawha Valley studies and in the Urban Air Toxics Program.

The results of the Denver and Staten Island studies were not available at this writing. Kanawha Valley, however, showed serious contamination problems developed under this method, invalidating the use of its samples. This is not to suggest that there is a general flaw with DNPH cartridges or HPLC analysis, but the Kanawha report does not discuss the cause of, or possible corrections for, this problem. This problem has been overcome in the Urban Air Toxics Monitoring Program by careful preparation of the sample matrix before use.

2.7. Evolving Monitoring Technologies

New developments in technology for analyzing air toxics offer the promise of increased breadth and depth for air toxic studies, with greater reliability and possibly lower costs. Two major EPA research efforts are providing measured data that can help guide present and future applied studies of the urban soup. These are the Total Exposure Assessment Methodology (TEAM) and the Integrated Air Cancer Project (IACP). In addition, two specialized monitoring techniques that were used in the studies under review are briefly discussed later in this section—the TAGA® and Remote Optical Sensor (ROSE) systems.

TEAM

The TEAM studies have collected extensive measured data sets of personal exposure and ambient concentrations of selected toxic air pollutants. These studies, of which the principal ones were conducted in New Jersey and California,[19] measured exposure of 20 toxic air pollutants to a total of over 600 subjects. The TEAM data base provides measured ambient and personal data to display more fully the range of personal exposures than can be revealed by the more traditional approach of using measured or modeled ambient concentrations as a surrogate for personal exposure. Chapter 4 more fully describes the exposure data collected in the TEAM studies. This section briefly discusses TEAM's measured data set.

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[19] Additional field studies have been conducted in North Dakota, North Carolina and, most recently, in Baltimore, Maryland, a study still in progress as of this writing. More studies are planned.

The TEAM study personal samples were collected by human subjects wearing vests containing a Tenax® cartridge and a sampling pump; samples are collected over two 12-hour periods for each subject.[20] Gas chromatography/mass spectrometry (GC/MS) have been used for analysis of 20 preselected compounds. In parallel with the personal sampling, each TEAM study took ambient samples using traditional fixed site samplers. In each case, approximately 20 liters of air were collected during the sampling period and the 12-hour integrated averages were compiled. Tenax® was used for both personal and ambient samples in the early TEAM studies. The recent Baltimore TEAM study employed a combination of Tenax® (for personal sampling) and canisters (for fixed stations).

During the TEAM studies, several criteria were used to select the target chemicals (EPA, 1986). These criteria included:

1. Toxicity, carcinogenicity, mutagenicity;
2. Production volume;
3. Presence in ambient air or drinking water;
4. Existence of National Bureau of Standards permeation standards; and
5. Amenability to collection on Tenax®.

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[20] Despite the problems with Tenax discussed by Walling (Walling 1984), TEAM studies continue to rely on Tenax for at least two reasons. First, Tenax appears to be acceptable for characterizing distributions of air concentrations, which is an important output of the TEAM studies; Tenax problems are most severe when the goal is to measure minor variations in concentrations over time, as is typical with fixed-station programs. Second, no satisfactory substitute for Tenax appears to be available for portable use in personal monitoring vests.

A total of 20 compounds have been routinely measured by TEAM, but 11 compounds were given particular emphasis during interpretation because: (1) this subset was identified as being the most amenable to the Tenax®/GC/MS sampling and analytical techniques, and (2) these pollutants were identified as being of greatest interest from a health perspective. The pollutants emphasized were:

1. Chloroform
2. 1,1,1-Trichloroethane
3. Benzene
4. Carbon tetrachloride
5. Trichloroethylene
6. Tetrachloroethylene
7. Styrene
8. meta or para-Dichlorobenzene
9. Ethylbenzene
10. o-Xylene
11. meta or para-Xylene.

The Integrated Air Cancer Project

Drawing on the combined expertise of four EPA research laboratories, the IACP program probably provides the best opportunity to further develop monitoring methods for toxic air pollutants. The broad nature of IACP, with the goal of identifying species most likely to be carcinogenic and their sources, requires development of monitoring methods to better identify specific carcinogens in a wide range of classes, including VOCs, semivolatile/particulate organic compounds, and inorganic pollutants. In addition to developing monitoring methods for directly characterizing pollutant species, IACP is also advocating the state of the art in monitoring methods for biological sampling and data collection to support receptor modeling applications.

Special Monitoring Techniques

This section briefly describes the TAGA® and ROSE® systems.

TAGA®

The TAGA® system is a mobile mass spectrometry/mass spectrometry (MS/MS) system that has been used to collect near real-time air quality concentrations. TAGA® is a mobile system that can be driven to many pre-selected sites, or used to more randomly search for "hot spots." Some limited sampling was performed in Santa Clara (EPA, 1984) and Denver (Dumdei, 1986) using this system over a one-week period in each study area.

Problems with detection limits lessened the ability of this system to meet the objectives of the Denver and Santa Clara studies—most compounds at most sites were present in the ambient air at levels well below the detection limits of the TAGA®, limiting sampling to major intersections during rush hour in Denver and some vent sampling in Santa Clara. Data gathered were not useful to support the broader objectives of these studies.

ROSE

The Remote Optical Sensing System (ROSE®) is a portable infrared sensor housed in a van. It was used to monitor at selected municipal and hazardous waste landfills in New Jersey as part of the Philadelphia study (EPA, 1986). As with the TAGA® system, high detection limits adversely affected meeting the goals of the study. The data were not usable for meeting the study's stated objectives.

2.8 Insights into the Use of Monitoring in Air Toxics Programs

Cost-Saving Measures

Since monitoring programs are often the most expensive component of an urban-scale air toxics study, they often offer the greatest potential for employing cost-saving measures. Perhaps the key point to consider when designing a cost-effective monitoring program is that the goal is to best characterize air quality, often for a metropolitan area.

Assuming funds are fixed, then tradeoffs exist between the coverage of the sampling program (i.e., the number of sites and days of sampling per site) and the data quality objective of each sample.

If cost-saving measures can be instituted to improve spatial and temporal coverage, and if the data quality of each sample can be documented to be within limits defined as acceptable to the project, methods such as those mentioned below may help improve spatial and temporal coverage.

Sample Compositing—Sample compositing (combining short-term samples before analysis to increase the temporal representativeness of each composited sample) can be an attractive concept, especially if the major concern is with carcinogenic risks, where annual averages are the most important statistic of concern. Although metals have been composited effectively for many years and the concept has been used with source testing, only during the past several years has compositing gained acceptance as a viable potential option for ambient air sampling.

Instead of combining short-term (e.g., ≤24 hr) samples after they are collected, a variation of sample compositing involves the collection of long-term, but intermittent, samples. An example of this might be for each sample to be collected over a period of several days or a week, but actually "pulling" sample air for only 15 minutes within each hour of sample duration. Thus, the sample compositing is being done, in effect, by the sampling device rather than by the analyst in the lab. One concern with this approach is the potential for pump failure, which could result in substantial loss in coverage. Another concern is the possibility for sample loss or degradation over the long collection period or during subsequent storage.

To date, sample compositing has only been used in laboratory testing and was not used in any of these studies, with the exception of semivolatile organic compounds in the Denver study. While further documentation appears to be needed to better describe the tradeoffs in terms of accuracy, especially for VOCs, sample compositing appears to offer the potential for making effective air toxics monitoring programs more affordable at the local level.

Indicator Pollutants—Another potential cost-saving measure is the use of indicator pollutants to priortize and/or limit subsequent sample analysis. The concept here is to identify an easily measured substance or property that reasonably relates to other substances or properties that are more expensive to measure. Thus, one can rank the samples by the value of the indicator pollutant to isolate a subset of the samples that warrants further analysis. In the Denver IEMP study, for example, total organic carbon will be used to rank semivolatile samples collected at each site. The groups of low, medium, and high concentrations based on total organic carbon will be analyzed in composited batches. This offers the benefit of performing individual analysis for the upper end of the distribution, which may be of greatest interest in terms of short-term exposures, and also provides a means of quality assurance of the compositing step.

Periodic Network Enhancement—This technique is based on the concept of a routinely operated core monitoring network (such as three sites for a metropolitan area), which can be expanded to a larger network (such as eight to ten sites for a metropolitan area) to collect very detailed data for a short time period every two to three years. Having a screening-level monitoring program with a larger network in place aids the selection of optimal sites to address local needs. Performing follow-up monitoring every two to three years for a one-season program could provide documentation concerning changes in the relationship between the core and supplemental sites. This approach provides the benefit of an expanded network to characterize spatial distributions, without the continuous collection of data at the supplemental sites and attendant costs. None of the studies have applied this concept to date.

Selective Analysis—Another approach to reduce the costs of monitoring programs is the concept of selective analysis. If samples can be collected at a fraction of the costs of analysis (as is the case for many methods), a larger data set can be collected than analyzed, i.e., every sample that is collected does not need to be analyzed. By this approach, samples can be selected to fill out a matrix of conditions of greatest importance to a study; for example, worst-case dispersion, flow from specific facilities, or nonprecipitation days. By having broad coverage, the objectives of a monitoring program may be achieved by selecting those samples that best meet prespecified objectives. If estimates of annual averages are to be based on user-selected days, however, the burden is on the user to demonstrate that the averages are not biased by selecting unrepresentative days. For example, a weighting scheme may be needed to effectively represent long-term averaging. The South Coast study, which is using a form of selective analysis for its monitoring program, could benefit by describing the representativeness of the measured data set to estimate annual average conditions.

The "Supersite" Concept—As part of the Philadelphia study, a quantitative methodology was developed to select optimal number and location of monitoring sites based on dispersion modeling of available emissions data (see Appendix B). The goal was to maximize the use of available information to select the core sites that best met project goals. The cost of air toxics monitoring is expensive, especially the cost of maintaining long-term sites that might be used to help track trends in urban air toxics levels. Allocating a fraction of these resources to help support the selection of sites that provide the most independent data is one way of obtaining the most information per monitoring dollar spent.

This approach can be briefly summarized as follows:

1. Model the available emissions data for the key pollutants to be addressed in the monitoring program. Perhaps 5 to 10 pollutants could be used to guide the selection of monitoring sites. Normalized modeling with tight resolution would be recommended, such as a 1 km grid spacing.
2. Based on the initial results, select a long list of 25 to 30 potential site areas (at the block or neighborhood level).
3. Compute the correlation of modeled hourly or daily ambient concentrations among all sites for each of the targeted pollutants based on meteorological and emissions data that are representative of seasons in which monitoring programs will be done.[21]

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[21] Consideration of summary statistics could also be included in consideration of independence.

4. Select a maximum correlation, such as a correlation coefficient in the range of 0.6 to 0.8. For purposes of illustration, consider 0.7 as the desired cutoff.

5. For each pollutant, group all sites that are correlated by 0.7 or higher as a cluster.

6. Develop a scoring technique that identifies sites that provide the most independent information. (Refer to Appendix A for the details of the proposed scoring technique.)

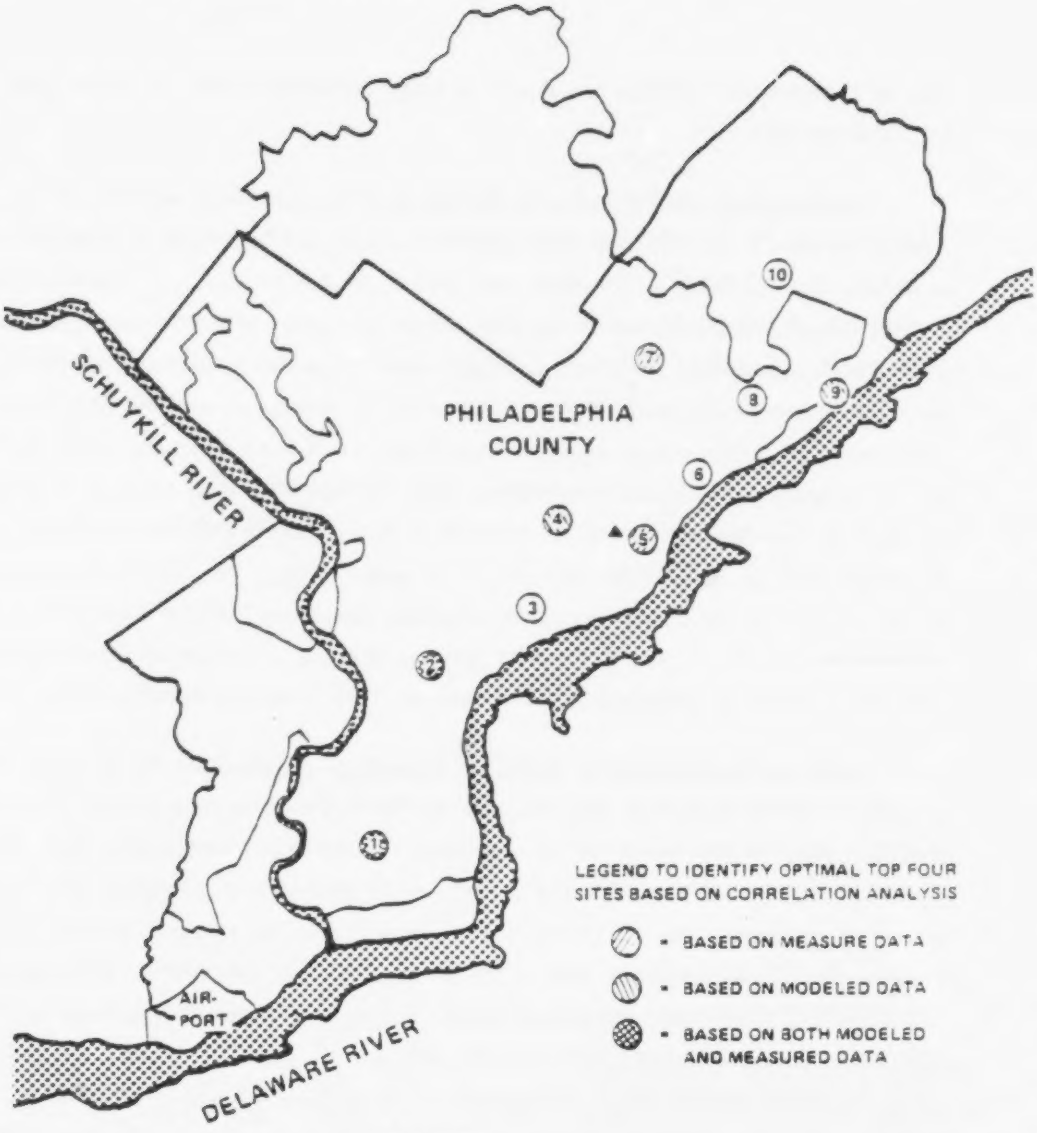
7. Develop an iterative procedure that eliminates sites that provide the least amount of information, until the only sites left are the most independent.

The Philadelphia Study presented the opportunity to test this approach by using this methodology for the actual monitoring sites, and then comparing the model-based results with rankings based on the actual measured data. Figure 2-2 shows the optimal four sites based on modeling and monitoring. As shown , there was overlap on two of the four sites. Appendix B provides a more detailed description of this potential cost saving measure.

General Issues

Accuracy, Precision, and Representativeness—It is a common misconception that measured air quality data are inherently "better" than modeled concentrations. As the experience of these studies shows, measured data can have as many limitations as modeled estimates of ambient air concentrations and can be equally misleading—more so, perhaps, because of their presumptive validity. Especially in air toxics studies, where costs tend to be high and funding limited, the number of sites and samples are often very small. Data sets developed to characterize urban-scale concentrations may often not be representative either in space or in time. In addition, the uncertainties of sampling and analytical methods, as well as the errors that can be present in identifying and quantifying pollutants, can often reduce the effectiveness of monitoring approaches.

This is not to say that monitoring should not be an important element of air toxics studies, but rather to suggest that monitoring should, in most cases, (1) be carefully planned in relation to its own study objectives and (2) be integrated as much as possible


PHILADELPHIA COUNTY
SCHUYKILL RIVER
DELAWARE RIVER
AIR-PORT
1
2
3
4
5
6
7
8
9
10
LEGEND TO IDENTIFY OPTIMAL TOP FOUR SITES BASED ON CORRELATION ANALYSIS
- BASED ON MEASURE DATA
- BASED ON MODELED DATA
- BASED ON BOTH MODELED AND MEASURED DATA

SAMPLING SITE

1. NAVAL HOSPITAL
2. GOODYEAR
3. FD 16
4. FD 7
5. ST. JOHN
6. LARDNER'S POINT
7. FD 71
8. FD 36
9. SAM BAXTER
10. NE AIRPORT

▲ ROADWAY METEOROLOGICAL/MONITORING SITE

FIGURE 2-2 OPTIMAL TOP FOUR MONITORING SITES FOR PHILADELPHIA PROGRAM BASED ON MODELED AND MEASURED DATA

into complementary analytic processes—namely, emissions inventory development and air dispersion modeling.

Vulnerability of Programs to Project Schedules—In a number of instances, monitoring results have been to some degree compromised by overall program schedules. Especially for estimating carcinogenic risk, where the key parameter of concern is annual average concentrations (either for the population as a whole or for the maximum exposed individual), monitoring programs of single seasons (or even shorter periods) must be carefully reviewed to ensure that the risks are not unduly biased by unrepresentative meteorological conditions or seasonal variations in emission rates from key sources or source categories. Ideally, concentrations should be estimated over a period of more than one full year, as has been routinely possible with the criteria pollutants. Unfortunately, air toxics funding tends to be provided on a special project basis, and few projects can afford more than selective seasonal sampling programs. Much has been done to compensate for this perennial problem through analysis of annual meteorological data, links to air modeling efforts, and the like, but the issue remains a significant one.

Lack of Comprehensive Pollutant Coverage—Limitations in funding, lack of suitable ambient sampling and analytic methods for some pollutants, and lack of available potency data mean that all monitoring studies must concentrate on a subset of potential pollutants of concern. The subset may be limited to carcinogens only, to certain classes of pollutants (metals, volatiles, etc.), or to a selected group of substances from a variety of classes; however, the effect is the same—a possible underestimation of exposures and risks from certain pollutants. For example, even studies such as IACP or the Denver study, where the goal is broad pollutant coverage, the limitations of the state of the art for measuring many compounds in the ambient air are substantial. Although measurement techniques continue to improve, only those compounds that have been validated for the selected methods can be accurately quantified.

Compromises Among Program Objectives—Most, if not all, of the studies reviewed were compelled to make compromises among multiple monitoring objectives, with varying effects on project results. In many cases, these compromises did not seriously hinder the quality or usefulness of study results, but in others they apparently have. There is a tendency to attempt to respond to as many pertinent questions as possible. Serious problems can occur when compromises prevent attainment of one or more primary goals.

Use of New Technologies—Studies that have attempted to apply evolving laboratory techniques to operational field studies have frequently failed to meet their original program objectives. The recurring theme is that the detection limits for which these technologies were developed may not be appropriate for field studies of ambient air. Although there is every need to evolve new laboratory techniques for operational studies, projects on limited budgets should be skeptical of applying innovative sampling and analysis techniques in the absence of concrete information on their performance (especially detection limits and interference and contamination problems), reliability, suitability to planned field conditions, and costs.

Spatial Coverage—Generally, measured data are collected to help meet the objective of characterizing concentrations within a large area. It is impractical, however, to equip a large number of sites to meet this goal because of the high cost of monitoring most noncriteria pollutants. The range in the network size among these studies was 1 to 13 sites in an urban area. For applied studies, therefore, the analyst must extrapolate data from a limited number of sampling points to represent a much larger area.

The extrapolation of measured data to a geographic area broader than specific monitoring sites needs to be carefully considered. For example, the 13-station network for the Staten Island study has by far the most extensive spatial coverage among the studies reviewed. The 13 sites, however, are a very small subset of the points in a study area that likely has large concentration gradients in some subsections. The question is where these specific sites fall within the spatial distribution of concentrations.

Dispersion modeling can help shed light on this problem, which again points to the need to more fully integrate monitoring and modeling components of such studies.

Dispersion modeling provides the opportunity in some cases to estimate likely contamination gradients throughout an urban area prior to establishing a monitoring network. Interpreting the modeled results, including evaluating exposure correlation among candidate monitoring sites, may improve the site selection process. Refer to Appendix B for further details.

Temporal Coverage—Ambient measurements will reflect tremendous variability as a function of time because of meteorological and emissions variability. This is most significant for major point source releases, as was shown in the measured data for Kanawha Valley where the industrial clusters dominated impacts for selected compounds. It was also shown in the Philadelphia IEMP study (for 1, 2 dichloropropane and 1, 2 dichloroethane), where a major wastewater treatment plant was a dominant emitter that could show two orders of magnitude difference in ambient concentrations as a function of wind flow alone. Area source-dominated receptors are less affected, but seasonal variability and diurnal patterns in emissions and meteorology will nevertheless produce substantial variations in concentrations.

For carcinogens, the goal in these studies was to estimate annual average concentrations, so the question becomes one of how well the short-term measurements for each site represent annual average conditions. The IEMP Philadelphia study made this comparison during a 30-day winter sampling program. Concentrations were modeled to match the days of the monitoring period, and also for a five-year climatological average data set. These modeled data were compared with the measured data set and found to be in reasonably close agreement.

In the studies reviewed, the sampling program ranged from 5 days per year in Santa Clara up to 50 days in Staten Island. Intuitively, it appears that Santa Clara data would be totally inappropriate for use in estimating annual average conditions. Staten Island, on the other hand, should reasonably represent annual average at these sites. Modeling similar to what was done for the Philadelphia study could help confirm this hypothesis.

Acute noncancer risks, because of their relation to short-term exposures, pose substantial challenges in terms of temporal coverage. Here, the problem involves characterizing peak concentrations, such as over a 1- to 24-hour period. None of studies to date provides sufficient measured data to effectively evaluate acute exposures.

Methods Limitations—It appears that methods for measuring toxic air pollutants are becoming more standardized over time, but there still are no standard methods such as those that exist for criteria pollutants. Until standardization is achieved, inherent variability in the data collected by different studies will persist. This may inadvertently lead to invalid comparisons of the resulting risk estimates across different metropolitan areas.

Chapter 2

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CHAPTER 3

EMISSION INVENTORIES

The following topics are discussed in this chapter:

- Use of emission inventories in multi-pollutant, multi-source urban air toxics assessments
- Pollutant coverage
- Source coverage
- Estimating emissions
- Spatial and temporal resolution
- Quality assurance
- Insights into compiling inventories for urban air toxics assessments

3.1 Use of Emission Inventories in Multi-Pollutant, Multi-Source Urban Air Toxics Assessments

Compiling an emission inventory is one of the first steps normally taken in urban air toxics assessment programs. Emission inventories can be used in many ways in air toxics programs. They can be used to identify sources and emission strengths, patterns, and trends. They can be used to store information from related programs, such as permit or right-to-know data. They can be used to predict ambient concentrations and to assist in the development of control strategies and regulations.

Most commonly, in urban air toxics assessment studies, emission inventories are used as input to ambient air dispersion models for estimating concentrations, exposures, and risks across an urban area. In the dispersion modeling approach, all inventoried sources within a study area are modeled, using one or several dispersion models to estimate pollutant concentrations across a representative receptor network. The resulting model-predicted concentrations are applied to a population distribution to yield estimates of the number of persons exposed to each concentration. The resulting person-concentration totals are then multiplied by cancer unit risk factors to estimate aggregate risk, which is a measure of the number of excess cancer cases expected in an area due to combined exposures to the pollutants of concern. Model-predicted concentrations can also be used to estimate individual risks, such an average areawide risk or risk to the maximum exposed individual. Another expression of risk could simply be the number of persons exposed to various levels of each pollutant.

A major advantage of the dispersion modeling approach is that it allows the agency to project changes in ambient air quality and risk as a function of projected changes in emissions. This allows the agency to test the impact of growth and alternative control measures on air quality and makes the emission inventory an important tool in control strategy development. A second advantage is that concentrations and risk can be estimated for many more receptors than could reasonably be covered in an ambient air monitoring network.

3.2 Pollutant Coverage

Table 3-1 shows the pollutants that were inventoried and modeled in the urban studies reviewed in this report. The fact that 70 compounds were covered in one or another of these studies is indicative of the difficult task confronting the study manager—he/she must choose from literally thousands of potential pollutants to define the coverage of the emission inventory, after which the inventory must be compiled for each of these compounds.

TABLE 3-1. Pollutants Inventoried in Urban Air Toxics Studies

POLLUTANT	SIX MONTHS STUDY (35 COUNTY)	SIX MONTHS STUDY (NESHAPS)	KANAWHA VALLEY IEMP	SANTA CLARA IEMP	PHILLY IEMP IEMP
ACETONE					
ACRYLAMIDE		X			
ACRYLONITRILE	X	X	X		
ALLYL CHLORIDE		X	X		
ARSENIC	X	X	X		
ASBESTOS					
BENZENE	X	X	X	X	X
BENZO(a)PYRENE	X		X	X	
BENZYL CHLORIDE		X			
BERYLLIUM	X	X	X		
BROMOCHLOROMETHANE				X	
BROMOFORM				X	
BUTADIENE, 1,3-	X	X	X		
CADMIUM	X	X	X		
CARBON TET	X	X	X	X	X
CFC-113				X	
CHLOROFORM	X	X	X	X	X
CHROMIUM(TOTAL)	X	X			
CHROMIUM+6					
COKE OVEN EMISSIONS	X	X			
DICHLOROBENZENE				X	
DICHLOROETHANE, 1,2-	(SEE EDC)	(SEE EDC)	(SEE EDC)	(SEE EDC)	(SEE EDC)
DICHLOROETHYLENE			X		
DICHLOROPROPANE, 1,2-					X
DIETHANOLAMINE		X			
DIMETHYLNITROSAMINE		X			
DIOCTYL PHTHALATE		X			
DIOXIN					
EPICHLOROHYDRIN		X			
ETHYL ACRYLATE		X			
ETHYL BENZENE					
ETHYLENE		X			
ETHYLENE DIBROMIDE(EDB)	X	X	X		
ETHYLENE DICHLORIDE(EDC)	X	X	X	X	X
ETHYLENE OXIDE		X	X	X	
FORMALDEHYDE	X	X	X		
GASOLINE VAPORS	X			X	X
GLYCOL ETHERS				X	
ISOPROPYL ALCOHOL				X	
ISOPROPYLIDENEDIPHENOL, 4,4-		X			

TABLE 3-1. (cont'd) Pollutants Inventoried in Urban Air Toxics Studies

POLLUTANT	SIX MONTHS STUDY (35 COUNTY)	SIX MONTHS STUDY (NESHAPS)	KANAWHA VALLEY IEMP	SANTA CLARA IEMP	PHILLY IEMP IEMP
LEAD					
MANGANESE					
MELAMINE		X			
MERCURY					
METHYL BROMIDE					
METHYL CHLORIDE		X			
METHYL CHLOROFORM					
METHYLENE CHLORIDE		X	X	X	X
METHYLENE DIANILINE, 4,4-		X			
NICKEL					
NICKEL SUBSULFIDE		X			
NITROBENZENE		X			
NITROSOMORPHOLINE		X			
PCBs		X			
PENTACHLOROPHENOL	X	X			
PERCHLOROETHYLENE	X	X	X	X	X
PHENOL				X	
POM/PIC	X			X	
PROPYLENE DICHLORIDE		X			
PROPYLENE OXIDE		X	X		
STYRENE	X	X			
TEREPHTHALIC ACID		X			
TITANIUM DIOXIDE		X			
TOLUENE				X	
TRICHLOROETHANE, 1,1,1-				X	
TRICHLOROETHYLENE	X	X	X	X	X
VINYL CHLORIDE	X	X	X	X	
VINYLIDENE CHLORIDE		X	X		
XYLENE ISOMERS				X	

TABLE 3-1. (cont'd) Pollutants Inventoried in Urban Air Toxics Studies

POLLUTANT	BALTIMORE IEMP	SE CHICAGO	FIVE CITY CONTROLLABILITY	SOUTH COAST MATES*	MOTOR VEHICLE
ACETONE		X			
ACRYLAMIDE		X			
ACRYLONITRILE		X			
ALLYL CHLORIDE					
ARSENIC	X	X	X	X	
ASBESTOS		X	X		X
BENZENE	X	X	X	X	X
BENZO(a)PYRENE	X		X		X
BENZYL CHLORIDE		X			
BERYLLIUM		X	X	X	
BROMOCHLOROMETHANE					
BROMOFORM					
BUTADIENE, 1,3-		X	X		X
CADMIUM	X	X	X	X	X
CARBON TET	X	X	X	X	
CFC-113					
CHLOROFORM	X	X	X	X	
CHROMIUM(TOTAL)	X	X	X	X	
CHROMIUM+6	X		X		
COKE OVEN EMISSIONS		X			
DICHLOROBENZENE					
DICHLOROETHANE, 1,2-	(SEE EDC)	(SEE EDC)	(SEE EDC)	(SEE EDC)	(SEE EDC)
DICHLOROETHYLENE					
DICHLOROPROPANE, 1,2-					
DIETHANOLAMINE		X			
DIMETHYLNITROSAMINE					
DIOCTYL PHTHALATE					
DIOXIN		X			
EPICHLOROHYDRIN		X			
ETHYL ACRYLATE		X			
ETHYL BENZENE	X				
ETHYLENE		X			X
ETHYLENE DIBROMIDE(EDB)	X	X		X	X
ETHYLENE DICHLORIDE(EDC)	X	X	X	X	
ETHYLENE OXIDE	X	X	X		
FORMALDEHYDE	X	X	X		X
GASOLINE VAPORS		X	X		X
GLYCOL ETHERS	X				
ISOPROPYL ALCOHOL					
ISOPROPYLIDENEDIPHENOL,					

TABLE 3-1. (cont'd) Pollutants Inventoried in Urban Air Toxics Studies

POLLUTANT	BALTIMORE IEMP	SE CHICAGO	FIVE CITY CONTROLLABILITY	SOUTH COAST MATES*	MOTOR VEHICLE
LEAD				X	
MANGANESE			X		
MELAMINE		X			
MERCURY		X	X	X	
METHYL BROMIDE					
METHYL CHLORIDE		X			
METHYL CHLOROFORM	X				
METHYLENE CHLORIDE	X	X	X	X	
METHYLENE DIANILINE, 4,4-					
NICKEL		X	X	X	
NICKEL SUBSULFIDE					
NITROBENZENE		X			
NITROSOMORPHOLINE					
PCBs		X			
PENTACHLOROPHENOL		X			
PERCHLOROETHYLENE	X	X	X	X	
PHENOL	X				
POM/PIC	X	X	X		X
PROPYLENE DICHLORIDE					
PROPYLENE OXIDE		X			
STYRENE		X			
TEREPHTHALIC ACID					
TITANIUM DIOXIDE		X			
TOLUENE	X	X		X	
TRICHLOROETHANE, 1,1,1-				X	
TRICHLOROETHYLENE	X	X	X	X	
VINYL CHLORIDE		X	X	X	
VINYLIDENE CHLORIDE		X			
XYLENE ISOMERS	X	X		X	

* South Coast actually included many compounds in their inventory not shown in this table. Only those pollutants included in their MATES are shown here.

Note in Table 3-1 that some compounds were inventoried in more studies than others. This gives some indication of what compounds might logically form a core list for consideration by a study manager at the outset of a new assessment. The following compounds were inventoried in five or more of the studies reviewed:

Arsenic
Benzene
B(a)P
Beryllium
Butadiene 1,3-
Cadmium
Carbon tetrachloride
Chloroform
Chromium (total or +6)
Ethylene dibromide
Ethylene dichloride
Gasoline vapors
Ethylene oxide
Formaldehyde
Methylene chloride
Perchloroethylene
POM
Trichloroethylene
Vinyl chloride

It is important to note that the above compounds account for the vast majority of aggregate cancer incidence in most of the studies reviewed. Of these pollutants, POM, formaldehyde, 1,3 butadiene, chromium, and benzene generally contributed most to aggregate incidence.

In addition, the above list of compounds conforms well with the recommended list of compounds given in Table 9 of earlier EPA guidance on compiling air toxics emission inventories (EPA, 1986), even though the lists were independently derived.

Some of the pollutants listed in Table 3-1 were inventoried out of concern for their contribution to maximum exposed individual (MEI) risks rather than to their contribution to areawide cancer incidence. Some pollutants were included because of their potential for noncancer health effects, while others were probably included because of general public concern.

The treatment of several pollutants contributing significantly to cancer risk—polycyclic organic matter (POM), hexavalent chromium (chromium +6), and formaldehyde—is important to note in the studies reviewed and discussed below. The treatment of beryllium and nickel is also discussed below, since, like chromium, the chemical form assumed in one's analysis will have an important effect on cancer incidence.

Treatment of Polycyclic Organic Matter (POM)

There was considerable divergence among the studies on the treatment of POM, which is sometimes (as in the Six Months Study) called PIC, or products of incomplete combustion. Two distinct approaches have been used in those studies where POM was addressed:

1. B(a)P Surrogate Approach. The Six Months Study, Southeast Chicago Study, and the Santa Clara IEMP assumed that all POM could be represented (with acknowledged uncertainty) by a surrogate compound, benzo(a)pyrene. The Six Months Study inventoried B(a)P, specifically, and then applied a potency factor that represented all PIC. Somewhat differently, the Southeast Chicago Study and Santa Clara IEMP inventoried all POM and then applied the potency factor for B(a)P. The Baltimore IEMP and Motor Vehicle Study also used this approach as one of several alternatives for assessing risk from POM.

2. Comparative Potency Factor Approach. This approach (described in greater detail in Chapter 6) applies a potency score (or cancer unit risk factor) to the entire mixture of POM emitted by each source category rather than to a particular surrogate compound. Comparative potency factors are forthcoming from EPA's Integrated Air Cancer Program and have been developed for road vehicle exhaust (diesel and gasoline fueled vehicles), wood smoke, and various other combustion sources. In the studies using comparative potency factors (Baltimore IEMP, 5 City Controllability, and Motor Vehicle), total solvent-extractable organic particulate was inventoried for those categories for which comparative potency factors existed. In these studies, POM was assumed to be represented by the solvent-extractable fraction of total particulate. As an option, total particulate can be inventoried rather than just the organic fraction if the comparative potency factors are increased accordingly to account for the non-solvent extractable particulate fraction that contains POM. The latter option allows the direct use of existing particulate emissions data.

The divergence that exists in the treatment of POM reflects the lack of consensus among study managers on which of these approaches (if any) is most suitable for estimating cancer risk. Hence, at present, POM incidence estimates must be considered to be highly uncertain.

Treatment of Chromium

Chromium is an important contributor to aggregate risk in the studies reviewed, but only the Cr^{+6} valence state is considered carcinogenic. Hence, the treatment of chromium is very important and has been approached in several ways. One approach is to treat all chrome emissions the same in the inventory (i.e., inventory total chromium) and then assume that some fraction of the resulting modeled ambient concentrations is Cr^{+6}. Early studies, such as the Six Months Study, very conservatively assumed that all chrome was Cr^{+6}, as did the South Coast MATES project in some of its risk estimates.. The Santa Clara IEMP assumed 10 percent of total chromium was Cr^{+6}, whereas the Baltimore IEMP assumed that varying fractions—from 0 to 100 percent—were Cr^{+6}.

The 5 City Controllability Study distinguished between Cr^{+6} and total chromium directly in the inventory and thus was able to model both ambient Cr^{+6} and total chromium levels. One advantage of this approach is that it allows one to project what fraction of ambient chromium is Cr^{+6}. This is important since Cr+6 cannot be directly measured at typical ambient levels.

Treatment of Beryllium and Nickel

As with chromium, it is important to be aware of the chemical form of other metals such as beryllium and nickel. In the 5 City Controllability Study, most beryllium was assumed to be in the oxide form rather than to be present in the more carcinogenic ore, fluoride, phosphate, or sulfate forms. Similarly, in this study, no nickel emissions were assumed to be in the carcinogenic refinery dust or subsulfide forms; hence, no cancer incidence was associated with nickel emissions. It appears from the various studies that conservatively assuming all beryllium and nickel emissions to be as carcinogenic as the most potent compounds containing these elements would greatly overestimate risks from these two substances.

Secondary Pollutants

None of the studies assessed secondarily formed (e.g., photochemical) pollutants via dispersion modeling since no validated models yet exist for predicting transformation products such as formaldehyde, peroxyacetyl nitrate (PAN), and the like. Hence, none of the studies expressly developed emission inventories of precursors of these secondary products for modeling purposes.

Generally, secondary pollutant levels have been approximated by using ambient air monitoring data. The 5 City Controllability Study, for example, used measured ambient air levels of formaldehyde in each city to estimate exposures to both primary formaldehyde emissions and photochemically formed formaldehyde. After backing out that fraction of ambient formaldehyde directly attributable to primary emissions, the balance was assumed to be due to secondarily formed formaldehyde. The 5 City Controllability Study then assumed, since formaldehyde is photochemically formed from VOC emissions, that each VOC emitter was culpable for that fraction of risk attributable to secondary formaldehyde in direct proportion to its level of VOC emissions.

3.3 Source Coverage

Because the emphasis in urban air assessments is on multi-source, multi-pollutant exposures and risks, most studies have tried to be comprehensive in their coverage of point, area, and mobile sources. So-called "nontraditional" sources have also been included in several studies, such as wastewater treatment facilities (including publicly owned treatment facilities or POTWs); treatment, storage, and disposal facilities (TSDFs), which include aeration tanks, landfills, and surface impoundments, for waste handling; waste oil combustion; hazardous waste combustion; and ground-water aeration. Most nontraditional sources involve pollutant transfer from another media (e.g., water or solid waste) to air and are not yet well characterized.

By and large, the sources covered in most urban air toxics assessments are the same sources covered in criteria pollutant inventories. However, most of the study managers were conscious of the need to inventory important sources of air toxics that may not have been included in their criteria pollutant inventories, because either (1) their criteria pollutant emissions were below some point source cutoff level or (2) they did not emit any criteria pollutants at all. Important examples of this kind of source include chrome platers, cooling towers (using chromium corrosion inhibitors), hospital sterilizers, wood stoves and fireplaces, and small surface coating/degreasing operations that emit toxic but photochemically nonreactive VOCs (e.g., methylene chloride).

Some sources received particular or unique emphasis in certain studies. The Santa Clara IEMP, for example, included drinking water treatment plants, municipal landfills, and semiconductor manufacturing in the inventory. The Kanawha Valley IEMP emphasized fugitive equipment and vent releases from the major organic chemical manufacturing plants. The Baltimore IEMP and Southeast Chicago Study both placed particular emphasis on large iron and steel facilities in each of their respective urban areas.

3.4 Estimating Emissions

Two approaches were used in the studies reviewed for estimating air toxics emissions. In one approach, emissions were derived from existing data bases. Alternatively, emissions data were developed from source-specific surveys. A combination of these approaches was used in most studies.

Emissions Estimation from Existing Data Bases

To differing degrees in the studies reviewed, existing data bases were used as a starting point for locating sources of air toxics and estimating emissions therefrom. Existing data bases included criteria pollutant inventories, published emission factors, species profiles, and information in the general literature. There were two variations on this approach. In one, an air toxics emission factor was applied to the existing throughput or activity level to estimate emissions. This works well for sources where the air toxics emission factors are expressed in the same units as the criteria pollutant factors (e.g., lb/ton of fuel burned or per vehicle mile traveled). All studies used this emission estimating approach for some sources.

A second variation is to apply species factors to existing VOC and particulate matter (PM) emission totals to estimate emissions for particular toxics. The Southeast Chicago Study provides an example of the use of the species factor method to estimate point source emissions. First, emissions estimates for total VOC and TSP were obtained from EPA's National Emissions Data System (NEDS) for each process within 88 point sources. Species factors (representing the fraction of the total TSP or VOC emitted as various individual species from each process) were then multiplied by the process-specific emission totals to yield emission levels of specific toxics. Information on species fractions are available from EPA. (EPA, 1988)

Source-Specific Survey Data

Local survey data were used in a number of the studies, including the 5 City Controllability Study, the Southeast Chicago Study, the South Coast Study and the IEMPs, to estimate the emissions of selected air toxics from point sources. The IEMPs illustrate the use of local survey data; with the exception of the Denver IEMP study, each study included the incorporation of survey-based emissions data into the emissions inventory used for the exposure assessment. Each survey was done somewhat differently, as shown below:

Philadelphia—The Philadelphia Air Management Services (PAMS) conducted a survey of 345 potential point sources of air toxics in the Philadelphia study area. Emissions estimates at the facility level were solicited from industry; these were, in turn, subjected to engineering review by the PAMS engineering staff. This survey did not, however, include a large wastewater treatment plant. IEMP estimated releases from this plant based on influent, effluent, and sludge data to help complement the PAMS inventory.

Baltimore—Surveys were conducted by the Maryland Air Management Administration (AMA) to obtain input data needed to estimate the emissions from approximately 200 point sources. AMA staff then computed emissions estimates at the facility level.

Kanawha Valley—The West Virginia Air Pollution Control Commission (WVAPCC) conducted a survey at the release point level for all major sources addressed in the study. WVAPCC staff conducted limited verification of these emissions data. A total of 17 facilities, 2,258 release points, and 570 pollutants were covered.

The WVAPCC provided guidance to the industries on generating emission rates for three major categories: process, combustion, and fugitive releases. Unlike the other studies, the Kanawha Valley study had very detailed release specifications, grouped into logical modeling units, (e.g., building sources, area sources, and stacks). The other studies did not subdivide the data below the facility level, resulting in relatively crude treatment of release specifications.

The vast majority of emissions within this study were from vents and fugitive releases rather than from stacks. Since the Kanawha Valley experiences such wide variations in meteorological conditions as a function of daytime or nighttime conditions, it was important to account for the variability in emissions on this basis to avoid introducing model bias for sources that primarily emit during the daytime hours. Hours of operation data were thus considered when using the inventory for dispersion modeling purposes.

Southeast Chicago—Questionnaires were sent to 29 of the 88 facilities in the source grid. These 29 were selected in a two-step process to rank the most important sources. First, all chemical manufacturing facilities were automatically listed because of presumed importance. Second, the remaining sources were ranked by probable impact of their VOC/PM emissions on the receptor area and on the probability of these sources emitting air toxics.

A questionnaire was sent to each of the 29 companies asking them to make their own emission estimates of about 50 pollutants and also asking for certain modeling data. Substantial follow-up was often necessary to obtain, clarify, or confirm company responses. In many cases, the companies asked EPA for species fractions to estimate their own emissions.

South Coast MATES—The South Coast Air Quality Management District (SCAQMD) conducted a mail survey of 1,606 companies in the South Coast Air Basin to update its toxics emission inventory. A number of local information systems were used to compile the emissions inventory. These information systems included SCAQMD's:

1. Automated Equipment Information System
2. Emissions Inventory System
3. Annual Emission Fee Reports
4. New Source Review Files.

Literature searches were conducted, letter mail-outs were sent, and telephone calls were made in cases where insufficient information was received.

3.5 Spatial and Temporal Resolution

Spatial resolution is a measure of how finely emissions data are subdivided (i.e., resolved) in space, whereas temporal resolution is a measure of how finely emissions data are subdivided in time. The resolution of any risk estimates resulting from an urban assessment cannot exceed the resolution of the emissions (and other) data used as input.

Grid Size and Grid Cell Resolution

Many of the studies superimposed a dispersion modeling grid over the urban area. Each grid comprised many, usually square, grid cells. The South Coast and Southeast Chicago studies did this as did the IEMPs in Baltimore, Kanawha Valley, Santa Clara and Philadelphia. The grid at once defines the receptor network for dispersion modeling (receptors being defined as one or several points within each grid cell) as well as the necessary resolution of the emission inventory.

Grid cell sizes varied from 1 km x 1 km in the South Coast MATES project to 5 km x 5 km in the Philadelphia and Baltimore IEMPs. (The Baltimore IEMP actually defined a "refined" grid, comprised of 2.5 km x 2.5 km grid cells over Baltimore City within the larger grid.) Southeast Chicago adopted a 2 km x 2 km grid cell size and the Kanawha

Valley IEMP used a 2.5 km x 2.5 km grid cell size. (Note that the IEMP studies also included special discrete receptors in residential areas near major facilities to support modeling of the maximally exposed individuals.)

Southeast Chicago actually defined separate source and receptor grids. Their source grid was 46 km x 46 km, comprised of 529 2 km x 2 km grid cells. The emission inventory was compiled for this area. In contrast, their receptor grid was only 13 km x 13 km and was comprised of 169 1 km x 1 km grid cells. The idea here was to define a smaller receptor grid within a larger source grid to assure coverage of all sources that could reasonably be expected to impact on the receptor grid. This same concept was also employed in the Baltimore and Philadelphia IEMPs. In the Southeast Chicago study, the receptor grid was located within the source grid, but skewed slightly off center in the direction of the prevailing winds.

Point Source Resolution

Typically, point source locations are known to the nearest 0.1 km in emission inventories, which is adequate resolution for areawide cancer assessments. Several of the studies (e.g., the Baltimore IEMP) may have lost a measure of point source resolution because the emissions data were submitted at the facility level and could not be assigned to specific stacks, vents, and the like, within the facility.

In studies evaluating maximum individual exposures and risks, more attention needs to be given to the actual release configuration within large, sprawling facilities that have different types of release points. The Kanawha Valley IEMP, of the studies reviewed, best delineated different kinds of releases within the large chemical manufacturing plants in the study area. Instead of assuming complex facilities could be characterized by one or several release points, up to 20 release groups were defined. Stacks were modeled individually. All vents on each building were modeled as a volume source based on the dimensions of the applicable building. Fugitive releases extending over an area, such as tank farms or equipment leaks, were modeled as industrial area sources. The Southeast Chicago Study also characterized several large coking operations at this level of resolution.

Area Source Allocation

All of the studies that used grid networks necessarily had to disaggregate area source emissions to the grid cell level. The 5 City Controllability Study did not define a modeling grid for each urban area, nor did it allocate emissions to the subcounty level. This is because EPA's Human Exposure Model (HEM)—used therein for dispersion modeling and risk assessment—does not require subcounty apportioned data. HEM (1) internally apportions county level area source emissions to the Block Group/Enumeration District (BG/ED) level based on the population within each BG/ED, (2) runs a simplified box model to calculate population exposure at each subcounty level, and (3) sums the resulting BG/ED risks to produce a measure of aggregate risk in the entire study area. HEM does not yield spatially disaggregated exposure and risk results as do the models used in the studies where gridding was done.

The basic allocation approach used in most of the inventories was to apportion county level emissions to individual grid cells based on some surrogate indicator(s) such as population, employment within certain SICs, or vehicle miles traveled (VMT). The inherent assumption is that area source emissions are distributed according to the known distribution of the surrogate indicator.

The simplest approach is to apportion all area source emissions by population. This was done in the South Coast MATES study for all area sources but road vehicles and service stations. Road vehicle emissions of toxics were distributed based on the known distribution of VOC in the criteria pollutant inventory. Service station emissions were clustered at street intersections based on a weighting model developed from a telephone survey.

In the Southeast Chicago study and the Baltimore, Santa Clara, and Philadelphia IEMPs, area source emissions were apportioned by a mix of surrogate parameters. In Baltimore and Philadelphia, digitized United States Geological Service (USGS) land use data were used, including residential land use, commercial/services land use, and industrial and transportation land use. As an example, dry-cleaning emissions were apportioned by commercial/service land use. As another example, Southeast Chicago used VMT distributions to apportion gasoline marketing emissions and used manufacturing employment to allocate degreasing emissions. In the Southeast Chicago study, special surveys were used to assess the potential locations of hospital sterilizers and chrome platers.

Temporal Allocation

Most of the studies appear to have collected annual emissions data with no consideration of diurnal, seasonal, or batch operation variability. This has generally been considered satisfactory for cancer assessments where the emphasis is on long-term exposures to pollutants.

Since the dispersion models used in urban assessments distinguish daytime and nighttime dispersion conditions, some error is introduced if emissions are assumed to be uniformly emitted both day and night. In the Kanawha Valley, Baltimore, and Philadelphia IEMPs, engineering estimates were made of diurnal emissions variability for certain source categories, and these estimates were then used as input to evaluate dispersion model performance (Sullivan, 1985a). This was done to reduce some of the bias introduced by assuming uniform emission rates throughout the day. The assumption of uniform emissions can result in a disproportionate amount of emissions being associated with the poor nighttime dispersion conditions.

3.6 Emission Inventory Quality Assurance

Quality assurance checks were not explicitly discussed in most of the studies reviewed, so for the most part the following discussion infers measures that were taken to assure emission inventory quality. Some of these measures may not have been done intentionally as quality assurance activities, but nevertheless served in that capacity.

Emission Inventory Review

A number of the studies incorporated reviews at various times to help assure that the best possible inventory procedures and data were being used. Ideally, such reviews would be done during the planning stages, at the end of the data collection effort, and when the results were compiled. As an example of this, the 5 City Controllability Study regularly reviewed source and emissions data coming out of EPA's regulatory programs and incorporated changes to reflect these data. Two particular EPA programs worth following are the NESHAPS program and the motor vehicle testing program. The NESHAPS program routinely generates source, emissions and risk data for specific facilities based on "114 letters," i.e., responses obtained under the authority of Section 114 of the Clean Air Act. As a result of following these programs, significant changes were made during the 5 City Study for several major source categories and pollutants. In addition, previously uninventoried sources and pollutants were uncovered and added.

Data Verification

In a number of the studies, including the Baltimore, Philadelphia, and Kanawha Valley IEMPs, State or local agency personnel spot-checked the adequacy of emissions data submitted by industry. In some cases, site inspections were made. Emissions estimates were reevaluated and revised or updated as appropriate.

The Kanawha Valley IEMP inventory was resolved at the stack/vent level of detail for large organic chemical manufacturing facilities. The State agency spot-checked release points that emitted large quantities of highly potent pollutants. For example, for the most important facility in the study area, the State requested backup calculations for 15 to 20 selected processes to confirm the emissions data.

In the Southeast Chicago Study, the emissions data for the largest point sources in the study area—several major coking operations—were personally reviewed by agency personnel who had inspected each facility.

Monitoring vs. Modeling Comparisons

The IEMP methodology (Sullivan, 1985b) is based on using measured data as a means of facilitating the quality control of emissions data. For many major facilities, especially where there are large uncertainties in fugitive and vent releases, it can be prohibitively expensive to perform comprehensive source testing. However, by measuring the composite plume, such as at points 500 to 1,000 m from the facilities, comparisons with modeled data can be an effective means of checking the reasonableness of the emissions data. This approach was used in designing the Kanawha Valley monitoring program.

Comparing model predictions with measured ambient levels was also done in the Philadelphia and Baltimore IEMPs to check emissions data. In one case, discrepancies led to significant increases in the inventory estimates for a large garment manufacturer. In another case, such discrepancies led to the identification of sewer influent and effluent lines to a major wastewater treatment plant as a missing source of toxic emissions. The IEMPs developed a computer program called the Monitoring and Data Analysis Module (MADAM), which facilitates this comparison of measured and model-predicted data. (MADAM is discussed further in Section 4.9.)

Receptor models represent another potential technique for quality assuring emission inventories, but have not been used for this purpose in any of the studies reviewed in this report.

3.7 Insights into Compiling Inventories for Urban Air Toxics Assessment

The following insights can be drawn from the studies conducted to date:

- The study area (grid) should be defined to be as large as possible to assure that all sources are included in the inventory that may impact on the proposed receptor sites. The Southeast Chicago study and Baltimore and Philadelphia IEMPs defined source grids larger than their respective receptor grids to account for all local sources impacting on the receptor sites. The Southeast Chicago study source grid was about 13 times larger in area than the receptor grid and accounted for "upwind" sources up to 20 km away in the direction of the prevailing wind. Theoretically, this approach should help account for local background levels of pollutants that are not secondarily formed or due to gradual global buildup.

- The chemical form of some pollutants is important in cancer risk assessments. For example, it is important to distinguish between total chromium and Cr+6; between total nickel and the carbonyl/subsulfide forms of nickel; and between the oxide and other forms of beryllium (particularly beryllium sulfate). Only some forms of these metals are considered carcinogenic. Limited test data are becoming available to allow this kind of speciation in emissions inventories.

- Consideration should be given to how POM will be handled in the inventory. Depending on the risk approach adopted in the urban study, emissions of either total POM emissions or some surrogate—probably B(a)P—will be required.

- Broad source coverage is important in urban air toxics assessments, as many sources contribute to aggregate risk. Area and road vehicle sources must be included for reasonable completeness. Chrome platers and cooling towers should be included, whether treated as area or point sources. Wood smoke should be included. Formaldehyde emitters—particularly road vehicles—should be included, even though aggregate risk from primary formaldehyde emissions is probably less than from secondary formaldehyde.

- The spatial relationship of release points within large, complex facilities (e.g., iron and steel plants or synthetic organic chemical manufacturing facilities) may need to be characterized if one's study will assess maximum individual risks near these facilities (< 1 km). This requires better delineation of stacks, vents, building dimensions, equipment leaks, and other fugitive sources than is typically available in most inventories without special site surveys. For broad screening of areawide aggregate cancer incidence, simpler point source emissions characterizations may be adequate.

- The spatial resolution of the area source inventory needs to be commensurate with the level of spatial resolution desired in the resulting risk estimates. To the extent possible, the study area manager should pick and choose from several socioeconomic and/or land use parameters as surrogates for allocating area source emissions rather than rely on a single indicator such as population. Use of the appropriate variables for different area source categories should improve the accuracy of the study results since the distribution of emissions within an urban area can have a great effect on population exposures.

- The use of EPA's Human Exposure Model (HEM) frees the study manager from having to allocate emissions to the subcounty level because HEM performs this step internally (down to the BG/ED level) based on U.S. Census Bureau data. A downside is that HEM only allows this subcounty allocation to be done by population or else a spatially uniform distribution is assumed, neither alternative being as accurate as the use of direct survey techniques and better surrogate apportioning variables.

- Limited data suggest that increasing the temporal resolution in the emissions data may improve the accuracy of the study results. Typically, most studies have just compiled annual average emissions. However, since emission rates differ considerably for some sources, both diurnally and seasonally, and since most models reflect different diurnal and seasonal dispersion patterns, it may behoove the study area manager to consider diurnal and/or seasonal patterns when compiling the emission inventory, especially if a more detailed assessment is contemplated or if short-term concentrations and maximum exposures are desired as output. No studies reviewed have attempted to compile short-term (e.g., hourly) variability in emissions data. This level of temporal resolution would probably not be warranted in an urban assessment whose focus was on cancer, but could be important for evaluating acute, noncancer effects.

- As with any inventory, data quality objectives and quality assurance are important. As a planning step, a formal data quality protocol should be considered that reflects all anticipated end uses of the emissions inventory. During and after inventory compilation, as many review steps as possible should be planned. These reviews should involve parties who will use the data, parties who might be affected by the study results, and parties who have particular expertise concerning certain aspects of the inventory.

- Comparing monitoring data with modeling results can provide insights into missing sources, missing pollutants, or erroneous emission estimates in the air toxics emission inventory.

Chapter 3

References

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CHAPTER 4

DISPERSION MODELING

The following subjects are covered in this chapter:

- Use of models for estimating exposure in multi-pollutant, multi-source urban assessments.
- Decisions affecting modeling protocols
- Model selection
- Release specifications
- Selection of receptor network
- Meteorological data
- Decay, transformation, deposition
- Model execution
- Model performance evaluation
- Insights

4.1 Use of Models in Estimating Exposure in Multi-Pollutant, Multi-Source Urban Assessments

Most studies of toxic air pollutants in the urban environment rely on dispersion modeling as their chief means of estimating ambient concentrations of pollutants. Modeling is used at scales ranging from microscale to urban-wide, sometimes including transport from neighboring regions. As shown by the studies under review here, concentration estimates are generally used directly as input to risk assessments, although they may also be used as a preliminary step in the planning of monitoring programs, as input to receptor modeling analyses, or, in conjunction with monitoring data, for the verification of emission inventories.[1]

The Gaussian models in use for toxic pollutant analysis were originally developed for analyzing ambient air quality for criteria pollutants and represent nearly two decades of practical development experience. There has been no need to develop new classes of dispersion models specifically for the noncriteria pollutants—Gaussian model performance has been demonstrated by White (White, 1984) and others to be effective for the annual or seasonal averages that are the most common averaging periods for air toxics studies. As discussed in this chapter, however, there are a variety of areas (e.g., dense gas releases) where the application of existing models to air toxics studies requires careful review and analysis to achieve best performance. How well a model performs for a specific use depends on the accuracy of emissions data, including release specifications; the representativeness of meteorological data for the area under study; and how well the application matches the source and terrain types for which the model was originally developed.

In the studies under review here, emission inputs were compiled for specific industrial facilities, area sources (such as mobile sources, commercial development, and residential heating), and a variety of miscellaneous sources sometimes referred to as "nontraditional sources," such as comfort cooling towers, volatilization of organics from sewage treatment plants, wood burning furnaces or stoves, and other small point sources not generally included (at least until recently) in urban emission inventories. Some studies relied predominantly on individually defined emission rates; most used default estimates for at least some categories of sources (see Chapter 3).

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[1] Potential future applications of dispersion models could include estimating deposition rates (to analyze indirect pathways of toxic chemical uptake), or analysis of hypothetical or actual accident scenarios.

Meteorological data are input to dispersion models to characterize the direction and speed of transport, estimate horizontal and vertical dispersion rates, and account for the vertical extent of turbulent mixing. For the modeling analyses of these studies, meteorological data sets were chosen that yielded annual average concentrations because this averaging period was needed for exposure assessments of cancer effects.

The advantages of the use of air dispersion modeling in urban scale assessments are straightforward. Modeling analyses generally require only a small fraction of the resources needed to conduct comprehensive ambient air quality monitoring programs. They can provide extensive spatial and temporal resolution of estimated concentrations and can cover pollutants for which suitable ambient air monitoring methods are not available. Modeling analyses can isolate the effect of any single source or evaluate the impact of any aggregate of sources. Finally, modeling can analyze hypothetical situations, such as the imposition of a range of control scenarios on existing sources, addition of new sources, or operational changes in the utilization patterns of facilities.

An important disadvantage of modeling lies in the limited validation of available models and the consequent uncertainties this produces. Available validation data apply to only a small subset of the scenarios in which models can be used, and models are often applied under conditions quite different from those for which they were developed. Also, models cannot accurately estimate concentrations unless sources are adequately characterized, which is problematic in many cases.

4.2 Decisions Affecting Modeling Protocols

Unlike monitoring programs, for which it is common (and often required) practice to prepare quality assurance/quality control (QA/QC) plans, modeling protocols have typically not been prepared to guide the design and execution of modeling programs. Ideally, all studies would benefit from defining protocols that would: (1) clearly define objectives to be achieved in relation to general program goals and (2) present the details of the approach to meet these objectives. Detailed elements of the approach would include:

- Level of analysis: In modeling, "screening level" or "scoping" analyses are often used as the first phase in a multi-stage study, such as to estimate the scope of potential problems or aid in design of monitoring programs. Depending on their purposes, screening level programs can use less detailed input data and more generalized modeling techniques. Where the purpose of a program is to develop refined modeling, input requirements are more stringent and modeling protocols are likely to be more elaborate.

- Source categories/emissions data: Study managers may elect to model only certain categories of sources (e.g., industrial point sources) or may attempt to be comprehensive, including a wide range of area sources and minor point sources. In addition, the level of detail in emissions data specifications may vary considerably from study to study. For instance, for screening purposes a major industrial source might be modeled as a single point, but for refined modeling might be treated as a combination of point sources, volume sources, and area sources.

- Pollutants to be modeled: Both gaseous and particulate pollutants may be of concern to air toxics studies. For most analyses, the design and execution of the modeling program is seldom significantly affected by pollutant selection. An exception would be where analysis of atmospheric reactions or transformations is required, but none of the studies reviewed here attempted such analysis.

- Receptor networks/Scale of analysis: Design of modeling programs is significantly affected by whether the scale of interest is to define average urban-wide pollutant concentrations or to define concentrations for the most exposed individual (MEI). These concerns strongly influence the design of the grid used to structure model outputs.

- Terrain: Terrain is a key factor of concern; modeling complexity increases significantly if terrain is defined as "complex"—i.e., if some receptor points in the study area are at higher elevation than release points, or if the study has unusual geographic features (proximity to large bodies of water, location in valleys or in mountainous areas, etc.).

- Meteorological data: The two factors of concern are the data's appropriateness and detail. Available recording stations for the region must adequately represent conditions in the study area. In addition, for uses such as running models in complex terrain, evaluating model performance, or estimating short-term concentrations, detailed hour-by-hour data may be required. In some cases, air toxics studies collected and used their own meteorological data set.

- Averaging period of interest: All the studies reviewed included consideration of carcinogenic pollutants, for which average annual exposures are the statistic of concern. Such studies can use more simplified data to achieve study goals. Modeling is also capable of estimating short-term exposures (e.g., exposures averaged over a few minutes to a few hours). Although few air toxics studies have investigated short-term exposures, short-term exposures may be of increasing interest in the future.

All of the modeling studies reviewed in this report dealt with these issues, even if they did not develop formal protocols. Table 4-1 summarizes their general goals, as reflected in published reports. Of the studies reviewed here, only the Kanawha Valley study prepared a formal modeling protocol covering the elements listed above. The Kanawha study was the exception because of the highly unusual geography of the study region; the Kanawha River lies in a deep, winding valley with steep walls, a situation that no single available EPA model is designed to model effectively. This study therefore developed a formal modeling and meteorological data collection protocol for review by the EPA Science Advisory Board to clarify the project's goals and document its methods (see discussion below).

A broader discussion of the above decision elements is given below.

Level of Analysis—Both the NESHAPS study and the 35 County study (within the general Six Months Study) can be considered screening level or scoping studies in their entirety. Several other studies used screening-level analysis for preliminary evaluations. The South Coast study conducted a screening-level evaluation of 4,200 grid cells to select sites for the monitoring program (see Chapter 2 for a full discussion). The Philadelphia study used a screening level analysis to help select study area boundaries. Major point sources across the general region, including sources in Delaware, were modeled using generalized data to see whether the effects of the Delaware sources were significant in the Philadelphia metropolitan area; their impacts were found to be negligible, so the Delaware sources were not included in the program. In Baltimore, screening-level modeling of VOC sources helped in the selection of monitoring sites. In the Kanawha study, simplified screening helped select pollutants and sources of greatest concern for refined modeling.[2]

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[2] Another type of screening, not involving air dispersion modeling, was used in the 35-County study: counties across the country were evaluated for inclusion in the study by ranking them (1) by total emissions and (2) by total emissions multiplied by population.

Table 4-1. Goals for Dispersion Modeling Analysis in Multi-Pollutant, Multi-Facility Air Quality Studies

(NOTE: See Chapter 1 for a summary of each of these studies.)

	Scales Addressed			
	Spatial Resolution (* included, ** emphasized)		Temporal Resolution (* included, ** emphasized)	
	Average Exposures	MEI Exposures	Annual Average	Short-Term Evaluations
Baltimore	**	*	*	
Santa Clara	**	*	*	
Kanawha Valley	*	**	*	††
NESHAPS†	**	*	*	
35 County Study†	**	*	*	
South Coast	**	*	*	
Southeast Chicago	**	*	*	
Philadelphia	**	*	*	Model Performance only
5-City Controllability	**	*	*	

† Components of the Six Months Study

†† Anticipated study.

Source Categories/Emissions Data—All the studies in Table 4-1 attempted to cover all categories of sources, both point and area. They varied, however, in the level of detail of their emissions statistics and release terms. As discussed further below, the Kanawha Valley study, for instance, developed highly disaggregated release points within the large chemical complexes of interest within the study; it also investigated diurnal differences in release terms, primarily because dispersion conditions within the valley sometimes vary dramatically from day to night.

Pollutants to be modeled: With the exception of the Kanawha study, none of the studies reviewed here evaluated pollutants that posed unusual technical problems in modeling. Where particulate matter was modeled, the implicit assumption was that particle size was less than or equal to 10 microns, and that gravitational settling was negligible and that all particulates could be modeled the same as gases. The Kanawha project investigated the possible effects of decay of pollutants within the modeling area: separate model runs were done to evaluate allyl chloride.[3]

Receptor Networks/Scale of Analysis—Most of the studies concentrated on developing estimates of population weighted average annual exposures. From the point of view of dispersion modeling, estimation of these average exposures poses the fewest technical problems and requires the most generalized types of data (e.g., average annual emissions and average annual meteorological data).

The Kanawha Valley study, and, to a lesser extent, the Philadelphia study, were the only studies that put relatively heavy emphasis on modeling of MEI (most exposed individual) exposures. The main objective in the Kanawha study was to determine whether maximum exposures in the neighborhoods of the large chemical complexes located along the 50 km valley corridor between Nitro and Belle are significant. It was also pertinent to focus the modeling effort on relatively near-field effects because of concerns, such as those expressed by the EPA Science Advisory Board, that modeling of broader exposed areas was technically questionable. The Philadelphia study modeled potential MEI exposures primarily because of concern in one heavily industrialized area of the city that air toxics exposures were high.

The primary technical difference in conducting MEI evaluations in these two studies had to do with resolution of the emission inventories. Because of the large size of the chemical complexes in the Kanawha Valley, several of which span many hundreds of acres, the study coded hundreds of individual release points (vents, stacks, storage tanks) within each complex so as to support detailed estimates of MEIs offsite.[4] The

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[3] Although this pollutant has a relatively short atmospheric half-life (about five hours) compared with others in the study, the effects of decay were determined to be negligible within the short transport distances of interest within the study area.

[4] Accommodating the large size of the source complexes also led to location of monitoring sites at between 1.5 and 5 km from the fencelines so as to ensure maximum coverage of multiple emission points (see discussion in Chapter 2).

Philadelphia emission inventory was not resolved at this level of detail, although some further detail was added for select facilities to support the goals of the MEI analyses.

Terrain—EPA modeling guidance stipulates that if any receptor points within a modeling area are higher in elevation than physical stack heights of any sources, then complex terrain models should be used to evaluate exposures for those locations. None of the studies, however, used complex terrain models for this purpose, even though the geography in several of the areas could be formally defined as "complex."[5] The reason for not including complex terrain modeling in the South Coast and Kanawha studies is that the influence of terrain on predicted concentrations would be minor for the distances and stack heights evaluated and, therefore, would not be critical to project objectives. Two complex terrain models were used, SHORTZ (Philadelphia Study) and LONGZ (Kanawha Valley Study), but terrain data were not input. These models were used because of the treatment of dispersion.

Meteorological Data—Most of the studies relied on available meteorological data, but three developed their own sets. The Philadelphia and Baltimore studies developed independent data sets, including sequential data for one-year periods, in order to better interpret their studies' monitoring data and to support model performance evaluation. The Kanawha Valley study developed its own data set because the closest available meteorological data station was located at an airport on a plateau above the valley, and was therefore entirely unrepresentative of conditions within the study area.

Averaging Periods of Interest—As noted, the risk evaluations conducted by these studies emphasized chronic health effects from long-term (i.e., annual or lifetime) exposures. The Philadelphia and Baltimore studies, however, used short-term modeling to aid in model performance evaluation. In addition, future phases of the Kanawha Valley study are planning the use of short-term exposure analysis for evaluation of possible health effects (such as respiratory or neurological impacts) of less-than-lifetime exposures to toxic air pollutants.

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[5] As discussed below, however, LONGZ—a complex terrain model—was used in several studies for other reasons.

4.3 Model Selection

The recommendations in EPA Guideline on Air Quality Models were used as a basis to select models for the Baltimore, Kanawha Valley and Southeast Chicago Studies. A combination of models (ISCLT and CDM) was used because none of the current models in the Guideline provide adequate treatment of complex industrial sources and urban-scale area sources. The NESHAPS Study, 35-County Study, South Coast Study, and 5 City Controllability Study (see Table 4-2), all used exposure models that contain a dispersion model that is not listed in the Guideline as a recommended model for this application. This does not imply that the model selection in these studies, or the Philadelphia Study, is not appropriate. A major advantage of using a Guideline recommended model, however, is that such a model has greater consistency among studies. Table 4-2 lists the dispersion models used in these studies.

Table 4-2. Dispersion Models Used in the Reviewed Studies

	Baltimore	Santa Clara	Kanawha Valley	NESHAPS	35-County	South Coast	Southeast Chicago	Phila-delphia	5 City Control.
ISCLT	X		X				X		
CDM	X						X	X	
SHORTZ								X	
LONGZ		X	X					X	
HEM-SHEAR				X					X
SCREAM						X			
GAMS					X				

Model Characteristics

Each of the models has specific strengths and limitations.

HEM/SHEAR—HEM (Human Exposure Model) is an exposure modeling system that internally includes national population data from the Census Bureau at the block group level[6] and national meteorological ("STAR") data from the National Climatic Center

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[6] Reference is made within this report to census data reported both at the block group/enumeration district (BG/ED) and block levels. For exposure evaluations,

(NCC). HEM was designed to accept output from any dispersion model that produces displays in a compatible format, although most urban air toxics studies using HEM have employed the internal SHEAR model rather than conducting modeling separate from HEM.

SHEAR (Systems Applications Human Exposure and Risk) is basically consistent in many respects with the stand-alone EPA dispersion models shown in Table 4-2; it uses similar treatments for dispersion coefficients, plume rise, and building downwash. Two major differences between SHEAR and the EPA-developed dispersion models are: (1) SHEAR uses simplified box model treatments and prototype sources to represent area sources (rather than using gridded area source data[7]) and (2) SHEAR does not contain a mixing height term.

HEM/SHEAR uses modeled data as a basis to interpolate average concentrations for all block groups within the modeling domain, such as from 30 to 50 km from a source. Contributions from multiple sources are summed to estimate total concentrations for population exposure within each block group. Since HEM/SHEAR uses national population data, it does not include the exact locations of residences within each block group, and can therefore develop only approximations of MEI exposures.

SCREAM—SCREAM (South Coast Risk and Exposure Assessment Model) is a version of HEM/SHEAR as developed for use by the South Coast Study. It involved the following changes to SHEAR:

1. The model's code was changed to facilitate the use of a 16-station meteorological data set that was available for the study area. The closest meteorological data set was selected when modeling each source.

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dispersion modeling usually uses population data resolved to the census tract level, apportioning that data to a modeling grid with cells that might typically be 1 to 5 km square. BG/EDs, used in both HEM and GAMS, are a smaller census reporting unit than a census tract. Concentration estimates produced by HEM and GAMS are used to estimate concentrations at the centroids of individual BG/EDs. The SCREAM model, which was adapted from HEM, considers the smallest census unit, i.e., block-level data.

[7] SHEAR uses prototype sources for some area source components, such as gasoline marketing. For example, rather than assume uniform emissions throughout a specified area, specific hypothetical sources are located on the modeling grid to match the expected density of the service station coverage.

2. Area source modeling was done using an adapted version of the Climatological Dispersion Model (CDM) rather than the box model treatment internal to SHEAR.

3. Population was resolved to the street block rather than the block group level.[8]

4. City-specific population growth projections were used.

The SCREAM computer model is being enhanced and will include options for population mobility, indoor versus outdoor exposure, and noninhalation routes of exposure. SCREAM will also be enhanced to provide better estimates of individual risk and community cancer burden in regions and subregions of the air basin. (Barcikowski, 1988)

GAMS—GAMS (GEMS Atmospheric Modeling System) is an exposure modeling system similar to HEM/SHEAR, developed by the EPA Office of Toxic Substances as part of its Graphical Exposure Modeling System (GEMS). It uses the ISCLT model for point sources and a box model approach for area sources based on national meteorological ("STAR") data.[9] GAMS integrates population data with concentration data in a similar manner to HEM, although the approaches differ for close-in receptors, which could lead to minor differences on this basis between the two models in exposure estimates across a study area.

ISCLT—ISCLT (Industrial Source Complex–Long Term) is designed to provide enhanced flexibility for evaluating complex industrial sources. It includes separate treatments for stack emissions, volume-source emissions, and area sources, as well as a wide range of control options to tailor the model run to match the degree of specificity in the available source and meteorological data. ISCLT requires joint frequency data (wind speed, wind direction and stability class data) in the "STAR" format. A prime limitation of ISCLT to support modeling goals in multi-pollutant, multi-facility studies is its relatively weak treatment of widespread urban area sources (e.g., mobile sources, residential heating, and distributed solvent use).

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[8] SCREAM used dispersion model estimates to predict concentrations at the centroid of the block or BG/ED, depending on the distance from the source.

[9] As noted below, at the time of the 35-County Study, GAMS used ATM as its internal air dispersion model, which tended to bias study results.

CDM—CDM (Climatological Dispersion Model) does not provide the degree of flexibility offered by ISCLT to address major point sources—which can be a significant limitation for estimating MEI exposures. Its greatest strength for modeling the urban soup is its relatively detailed area source treatment. If matched with highly resolved emissions data, for area sources, this model theoretically can provide the most representative treatment available for area sources. If MEI analyses are not required, the resolution in CDM may be suitable for most point or area source treatments. CDM requires meteorological data similar to ISCLT, except that neutral stability is subdivided into daytime and nighttime periods.

SHORTZ/LONGZ—SHORTZ and LONGZ are EPA-recommended second-level screening models for estimating short- and long-term averages, respectively, in urban areas having complex terrain. They have two potentially useful features: (1) representative treatment of urban area sources and (2) the potential to use site-specific measured turbulence data to characterize dispersion rates. This model requires meteorological data in the form of a joint frequency distribution as in ISCLT. The user has the option of using site-specific meteorological data to characterize dispersion rates, however.

Rationale for Model Selection

Baltimore and Southeast Chicago—These studies used a combination of two models to reach their objectives: ISCLT (for its strength in modeling complex point sources) and CDM (because of its relatively refined treatment of area sources). Model selection was guided by adherence to the *Guideline on Air Quality Models* (revised) (EPA, 1986).

Philadelphia and Santa Clara—As the first IEMP pilot study, the Philadelphia study tested two modeling approaches. The CDM model was used early in the study for all point and area sources, but the program later shifted to SHORTZ/LONGZ, primarily to have the option of using site-specific meteorological data (collected over a one year period to document the project's 10-station air quality monitoring network) to define dispersion conditions. SHORTZ/LONGZ were not selected because of their complex terrain capability.[10]

Santa Clara used LONGZ primarily for consistency with the concurrent Philadelphia study. During the initial design stages of the Santa Clara study, it was unclear if site-specific meteorological monitoring would be done as in Philadelphia. If a comparable meteorological monitoring program were done in Santa Clara, LONGZ would have provided the option of inputting this data set as an alternative means of characterizing dispersion. Such a meteorological data set was not collected, however.

Although Santa Clara has substantial terrain variations within the study area, the terrain features of LONGZ were not used, essentially because none of the releases in this study area were from stacks (process vents and fugitive releases dominated the inventory) and terrain rise consequently was not a sensitive model input.[11]

Kanawha Valley—Model selection for Kanawha Valley was controversial because of the region's complex topographical setting. At the outset, the study made the following assumptions to simplify the modeling analysis and model selection:

- The modeling region was confined to the valley floor, thereby removing the complication of selecting models recommended for use in complex terrain.

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[10] The Philadelphia Study might have been strengthened by a more detailed comparison of these two alternative modeling techniques. Although minor improvements were observed for a few pollutants in the model performance tests using the more specific dispersion treatment in SHORTZ (as opposed to the default stability class approach used by CDM), the benefits of using SHORTZ/LONGZ over the more simplified CDM approach were not established. Perhaps the most useful finding of the Philadelphia modeling analysis was that the use of the more specific evaluations possible with SHORTZ/LONGZ are best suited for settings with complex topography and for all applications where short-term averaging is a modeling goal.

[11] In retrospect, in the authors' opinion, an alternative modeling procedure, such as shown for Baltimore and Southeast Chicago, may have better served the needs of the Santa Clara Study.

- The valley was subdivided into four relatively homogeneous zones, each to be modeled independently to minimize complications in flow analysis.
- Modeling objectives emphasized MEI concentrations in the areas near the major chemical facilities; this was expected to simplify the influence of complex flows and minimize the need to consider reflection of plumes off the valley walls.

Based on these simplifications, the project selected a reference model (ISCLT), consistent with EPA's *Guideline on Air Quality Models* (revised) (EPA, 1986), and an alternative model (LONGZ), essentially for research purposes. The study's hypothesis was that ISCLT, which uses generalized stability classes, may not adequately characterize site-specific dispersion because of the Valley's complicated flow patterns. It therefore used LONGZ, which can accept measured site-specific turbulence data, as an alternative model to ISCLT and compared the results.[12] Overall, ISCLT met the project's objectives better—LONGZ substantially underestimated concentrations relative to the limited measured data set, usually by a factor of 2 to 3.[13]

Because of ISCLT's simplified method for handling emissions from large area source emissions, the study also used a third model—a simple box model treatment—to evaluate area source impacts.

5 City Controllability Study; NESHAPS Study— Several studies chose to utilize EPA's Human Exposure Model (HEM) because (1) it is consistent with what EPA has used in its NESHAPS regulatory program and (2) it currently includes dispersion modeling

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[12] LONGZ was used as an alternative model because it is not recommended within the EPA Guideline on Air Quality Models as a recommended model for this application. In this sense, it was used for research purposes.

[13] Although LONGZ showed a consistent bias in its estimates, it also showed systematic differences in dispersion rates caused by site-specific turbulent intensity differences, as hypothesized by the study. There appeared to be a benefit in using LONGZ or SHORTZ if concentration bias could be removed. It appears that the empirical algorithms used by both LONGZ and SHORTZ to relate turbulence to dispersion rates work best with elevated sources. Most of the industrial sources of concern in this study, however, had low-level release points.
The study concluded that the apparent bias in LONGZ's estimates could be removed by modifications to the model code; currently proposed investigations of short-term ambient conditions in the Valley may therefore use a suitably modified version of SHORTZ to exploit available site-specific turbulent intensity data.

exposure and risk characterization modules, all within one system. (HEM is being updated by EPA to give it much more capability.)

The NESHAPS and 5 City Controllability Studies used the HEM/SHEAR model. The use of HEM/SHEAR appears to have met study goals, but the limitation of no mixing height treatment may act to significantly underestimate average concentrations for some applications. It is also unclear if different conclusions may have been reached if EPA recommended dispersion models were used in lieu of HEM.

South Coast MATES—The South Coast adapted HEM/SHEAR for its own purposes by incorporating a modified version of the CDM model in place of SHEAR, but retained some of the assumptions of HEM/SHEAR, such as not considering mixing height in the calculations (Liu, 1987). The adapted model was named "SCREAM." The motivation to develop an alternative model appears to have been: (1) to conduct more sophisticated modeling techniques for area sources, (2) to take advantage of the extensive coverage of meteorological data in the study area, and (3) to obtain finer resolution in predicted concentrations. Development of the SCREAM model appears to have produced a model that better met the goals of the South Coast study, while retaining much of the exposure modeling capability available within HEM.

35-County Study—Model selection was difficult for this study. An approach was needed that could readily model the 600-plus point source and 35 metropolitan-scale area source treatments that make up this study. The EPA GAMS exposure model, developed by the Office of Toxic Substances, was selected over HEM primarily because: (1) model execution for this scale study was expected to be facilitated by the GAMS approach and (2) the analysts were more familiar with GAMS. The tight project schedule resulted in the selection of GAMS.

GAMS was successful in meeting the immediate objectives of this project, and appears to have provided more efficient execution and processing of this large data set than would have been possible with HEM. A limitation of this approach, however, observed two years after the completion of this project (Sullivan and Hlinka, 1986), was errors in the ATM model, which was the only Gaussian model available in GAMS at the time. An error in ATM's plume rise term, and questionable assumptions regarding depletion of mass from the plumes caused by wet and dry deposition (Sullivan, 1986), were hypothesized to underestimate toxic air pollutant risk throughout the 35 counties analyzed in the study.[14] (Sullivan and Hlinka, 1986) The magnitude of the underestimates is mainly a function of the exhaust temperature. For sources with high exhaust temperatures, (e.g., greater than 500° F), model underestimates could be in the range of a factor of 3 – 10, in the authors' opinion; at near ambient temperatures, underestimates might be expected to be low by a factor of 2 –3.

4.4 Release Specifications for Emissions

Model outputs can, in some cases, be sensitive to the degree of resolution in release specifications,[15] with optimal resolution being a function of the scale of analysis. For example, complex facilities can be represented as a single release point to estimate average exposures across a study area without necessarily sacrificing the representativeness of the average modeled concentrations. On the other hand, the level of detail in release specifications can be an important model input for MEI analyses—the greatest differences in modeled concentrations occur within the first 1 to 2 kilometers of a source, where MEIs are likely to be located.[16]

With the exception of Kanawha Valley, and to a limited extent the Philadelphia study, all modeling analyses used simplified treatments to characterize release

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[14] This problem points to the risk taken when selecting a dispersion model that has not been subject to the review process of recommended models in the *Guideline on Air Quality Models* (EPA, 1986). The same risk appears to be present in using the SHEAR model routinely used within the HEM system.

[15] Release specifications required as model input are as follows:

Stack Source	release height, inner stack diameter, exit velocity, and exhaust temperature.
Area Source	horizontal dimensions of area, characteristic release height.
Building Source	vent specifications (similar to stack data), dimensions horizontally and vertically of building (or structure).

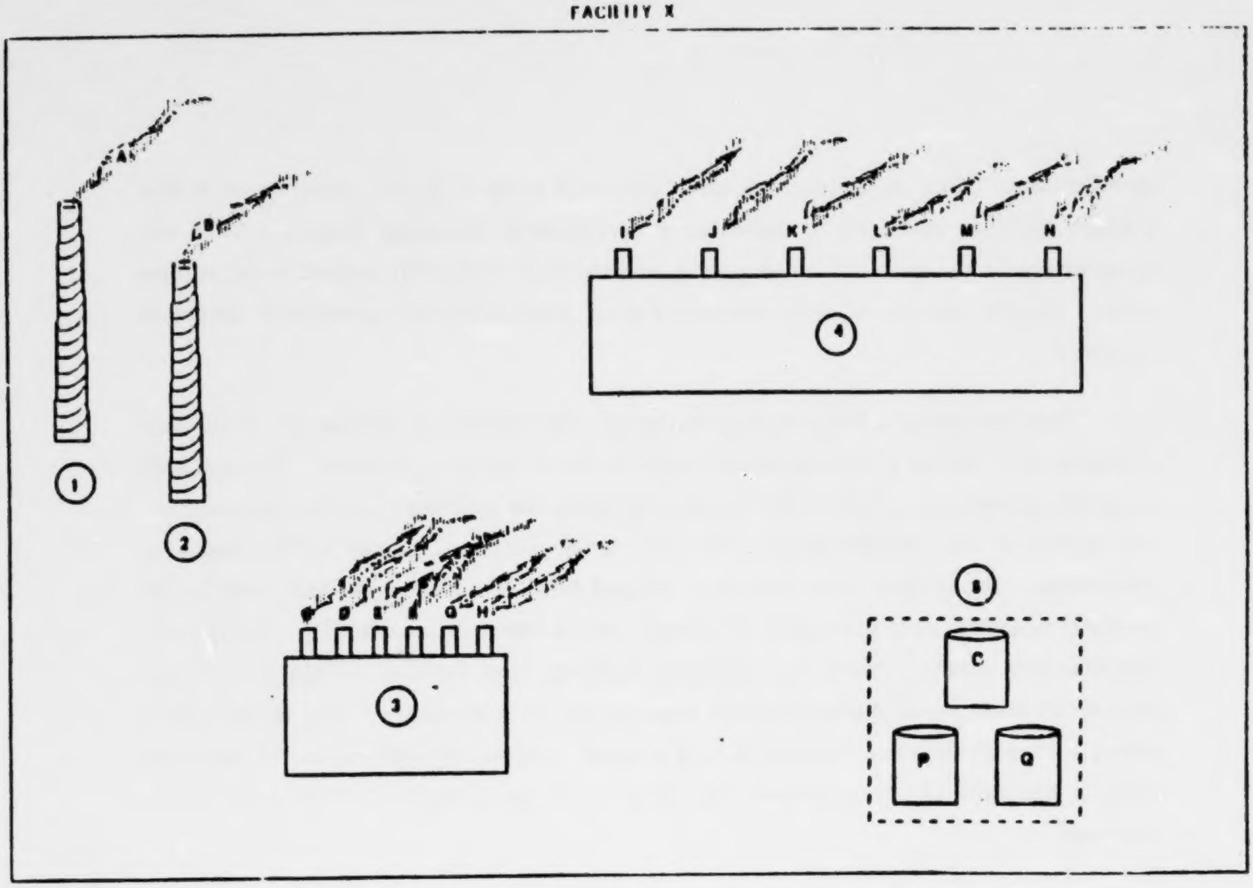
[16] This is especially true of air toxics evaluations, where most sources of concern are low-level sources that produce MEI concentrations relatively close to the facility itself.

specifications. Often an assumed release height, such as 10 m, or one release point within a major facility, was used to represent a complicated industrial facility. This was consistent with the goals of the modeling analyses (see Table 4-1) because most studies were primarily concerned with average annual population exposures and areawide incidence.

The Kanawha Valley study, however, was primarily concerned with MEI exposures and therefore needed greater detail in its emissions specificity. The available emissions inventory, compiled by the West Virginia Air Pollution Control Commission, was coded at the release point level—too much detail to model within resource constraints. The releases were therefore grouped into units that were more suitable for practical modeling purposes—up to 20 groups within each complex facility. Stacks were modeled individually. Vents on a common building were modeled within one volume source. In some cases, fugitive sources emitted over an area, such as tank farms, piping arrays, and so forth, were modeled as area sources. Figure 4-1 summarizes the approach used in Kanawha to group sources; this may be of use to future studies with similar emphasis.

4.5 Selection of Receptor Network Array

The receptor arrays used in these studies can be grouped into four categories: (1) grid-based selection of receptors, (2) a Block Group/Enumeration District (BG/ED)-based approach, (3) use of special receptors to match monitoring sites, and (4) complex terrain receptors.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Q

Group	Source Type	Specific Release Points
1	STACK	A
2	STACK	B
3	VOLUME SOURCE (BUILDING)	C,D,E,F,G,H
4	VOLUME SOURCE (BUILDING)	I,J,K,L,M,N,
5	AREA SOURCE	O,P,Q

Figure 4-1. Example of source grouping technique used for the Kanawha Valley Study.

Grid-Based Approach—In a grid-based receptor array, estimates of impacts from all sources modeled are output to a single set of cells, with each cell including one or more discrete receptor points within or along its boundaries. Since the grid is rectangular, all cells are the same size unless otherwise defined (some studies define subdivided cells in more densely populated areas). Figure 4-2 provides an example of a rectangular grid system used to define a receptor array.

This approach was used for the IEMP studies, where a rectangular grid was employed for all but the microscale analyses. First, a standard grid spacing was established (such as a 2.5 to 5 km square[17]). Average concentrations were then estimated for each grid cell by placing evenly spaced receptors (one to four per cell) within each cell and averaging all receptors within a cell to represent the average concentrations for all residents of the grid cell. Population within each grid cell was allocated by overlaying census maps on the grid and allocating population from each census tract to the grid proportionally on the basis of area (i.e., if half the area of a census tract lay within a cell, half the population of that tract is assigned to that cell). The population thus assigned is assumed to experience exposures equal to the average concentrations estimated for that cell.

Resolution tighter than 2.5 to 5 km grids is often needed to meet objectives other than estimating average concentrations. For example, the South Coast MATES used a 1 km grid system to help select monitoring sites in the South Coast Air Basin. As another example, the Southeast Chicago Study also used a 1 km grid system, albeit for only a relatively small (13 km x 13 km) receptor area. Within the IEMP studies, supplemental receptors within certain grid cells were needed to estimate MEI concentrations—the standard grid was complemented with specially selected locations (discrete receptors) that

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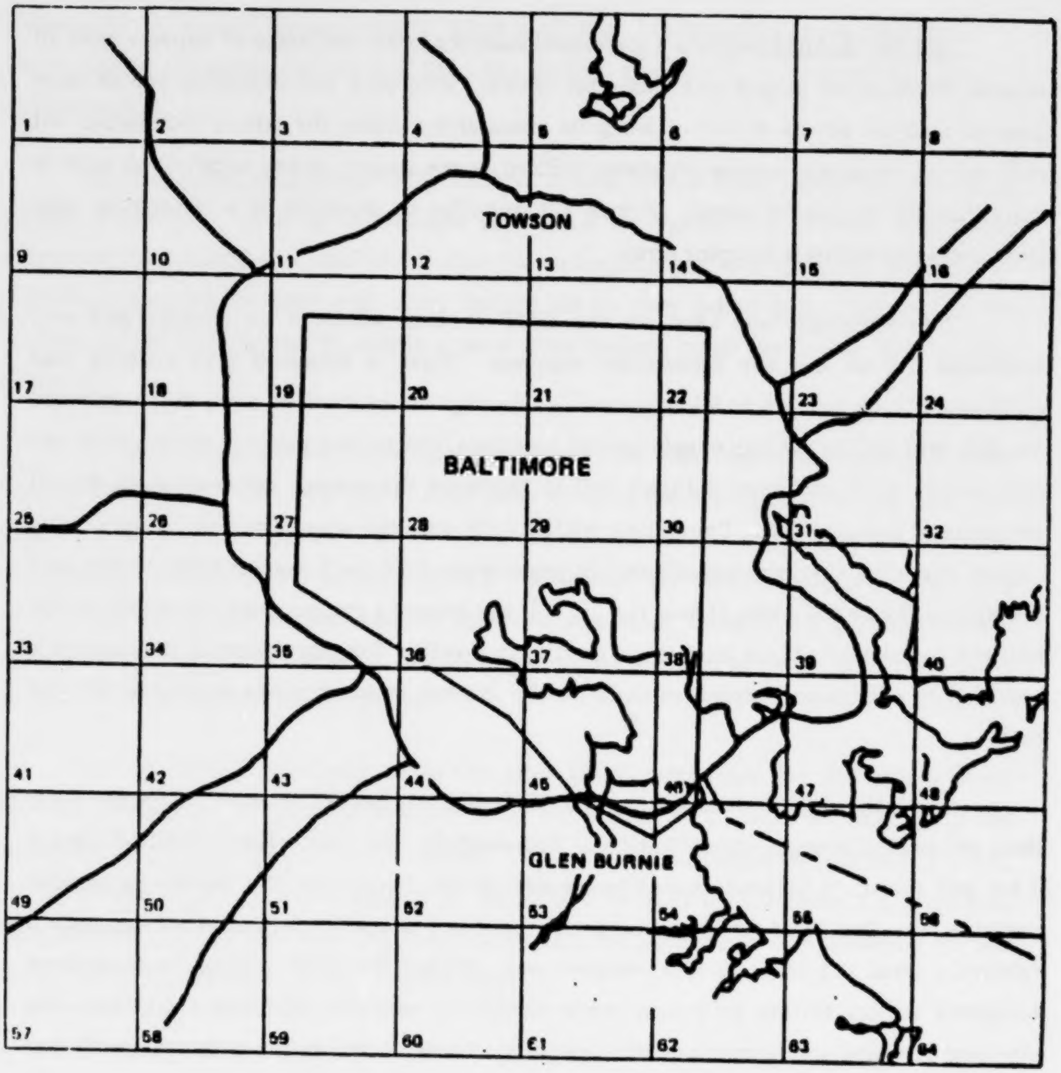
[17] For core urban areas with relatively high population density, or sharp gradients in concentrations caused by major industrial releases, a tighter (2.5 km) grid was generally used.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GLEN BURNIE
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Figure 4-2. Example of rectangular grid system.

represented residential areas closest to major facilities. The maximum concentration within residential areas was selected from the discrete receptor array to represent MEI concentrations.

BG/ED-Based Approach—This method is used in the HEM and GAMS exposure modeling systems. A polar coordinate grid (such as Figure 4-3) is used to estimate concentrations around each facility. Unlike the rectangular grid-cell approach in which all facilities output to the same grid, every facility has its own (polar) grid. Also unlike the rectangular grid-cell approach, cells in a polar array become larger the farther they lie from a source.

To aggregate exposures from multiple sources, the HEM and GAMS systems assign concentrations from each grid cell of each polar array to the appropriate BG/ED, and then aggregate exposures for each BG/ED.[18] HEM and GAMS use somewhat different methods of assigning concentrations to each BG/ED. In GAMS, all BG/EDs whose centroids lie within a particular cell are assigned the same concentration. HEM interpolates values and assigns interpolated values to each BG/ED based on the location of its centroid in relation to each cell's assigned receptor points.

An innovative approach along the same lines, developed for the South Coast MATES Study, was to consider distance from the source as a means of identifying the resolution to be used for receptor coverage. For example, for distances less than 2.5 km, the street block level of detail was used. BG/EDs were used for distances 2.6 to 10 km. This permits the model to allocate concentrations more accurately, compensating for the variation in grid-cell size across the polar array.

The HEM and GAMS exposure modeling systems are quite similar in their treatment of receptors, although there are some differences between the two in the treatment of receptors within 3.5 km of a facility. Because of its finer detail, the BG/ED

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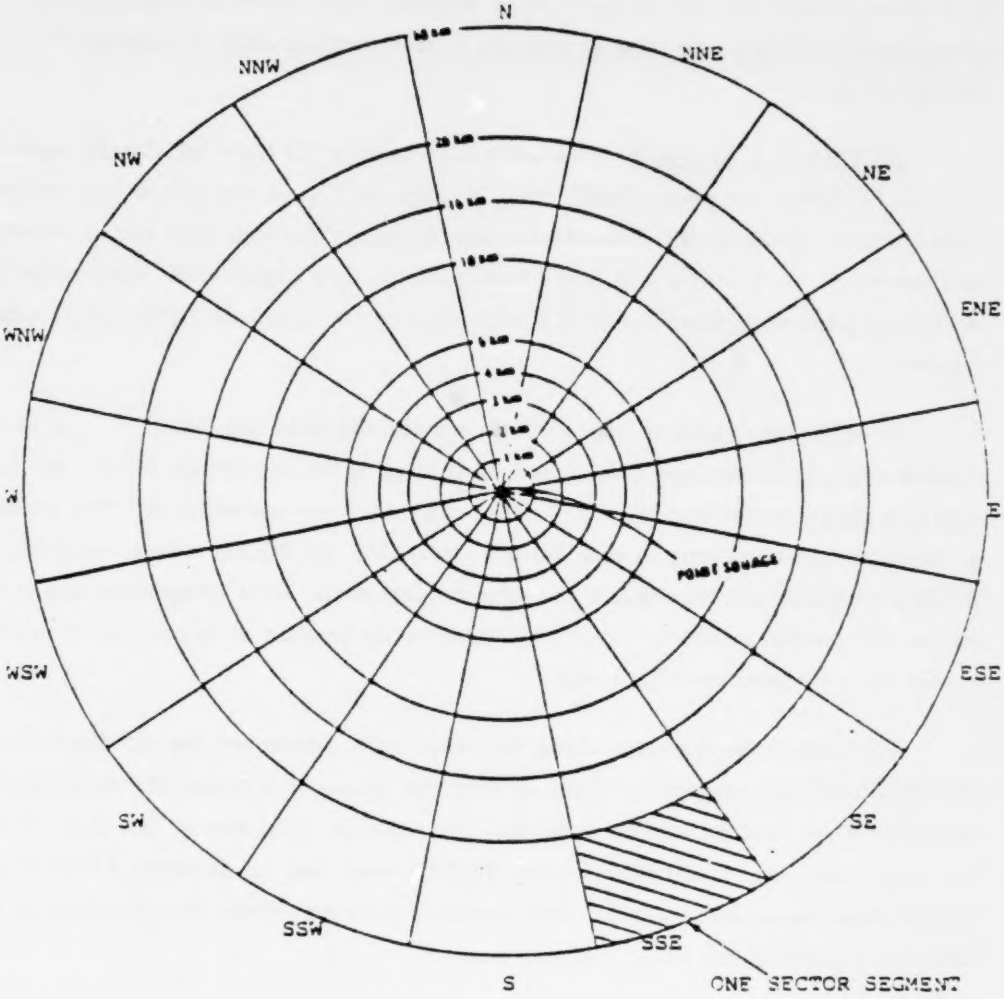
[18] A block group (BG) is an area representing a combination of contiguous blocks having an average population of about 1,100. Enumeration districts (ED) are areas containing an average of about 800 people and are used when block groups are not defined.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SSE
S
SSW
SW
WSW
W
WNW
NW
NNW
50 km
20 km
15 km
10 km
5 km
4 km
3 km
2 km
1 km
POINT SOURCE
ONE SECTOR SEGMENT

Figure 4-3. Example of Polar Grid System

approach should theoretically develop superior characterizations of average exposures for each cell than the rectangular grid approach, possibly leading to minor differences in estimates of total cancer incidence across a study area. Intuitively, it does not appear, however, that their more refined resolution in population data necessarily improves the accuracy of the exposure assessment on the metropolitan scale. Since the goal is to estimate total exposure generally across a metropolitan area (e.g., 50 x 50 km), minor differences in allocating population to model receptors would not be expected to make major differences in exposure. Differences at the fine scale are smoothed when producing an estimate of total exposure. The major limitation of these two exposure modeling systems is their relatively poor resolution for MEI exposures. Supplemental receptor coverage to include points representing distances to property boundaries or nearest residences appears to be needed when using HEM or GAMS if MEI analyses are one of the modeling goals of a study. (Refer to Chapter 5 for more detail on exposure assessment issues.)

Special Receptors Matched to Monitoring Sites—To evaluate the performance of models, the IEMP studies and the South Coast Study identified additional receptor points representing the locations of monitoring sites in order to be able to compare monitored ambient concentrations with modeled concentrations. Similar special sites are located where the study has exact knowledge of potential MEI locations, such as the exact distances to the fenceline of houses nearest a major source.

Selecting receptors for this purpose is obviously straightforward; the important concern here is that the exact location of the site be entered into the modeling analysis in order to accurately define the receptor locations. Exact locations are especially important when large point sources exist within 1 to 2 km of the monitoring or receptor sites.

Characterizing Terrain Rise for Receptors—Modeling analyses should include consideration of terrain differences between release heights and receptor elevations whenever terrain rise is expected to be a significant term in a modeling analysis. This can occur with complex terrain or when the effective height of the plume is substantially lowered with respect to the ground level. For emissions from elevated stacks, such as incinerators or power plants, terrain differences can be a significant issue.

The majority of the air toxics emissions considered in these studies, however, were from sources near ground level. Consideration of terrain differences between sources and receptors, therefore, was generally not a sensitive issue, and none of these studies incorporated it.

Of all the studies reviewed, the South Coast and Kanawha Valley projects had the greatest potential for terrain-related modeling complications. Both studies considered the issue, but concluded that it was unimportant in meeting project goals. In the South Coast study, power plants were the source category expected to be most affected by terrain, but these sources turned out to be located 10 km or more from areas with major terrain rise (Shikiya, 1987). In Kanawha Valley, flat terrain modeling was used even though large terrain rise occurred within 1 km of the sources because: (1) in nearly all cases, pollutants were emitted from near ground level; (2) the modeling goals emphasized MEI concentrations, which were essentially unaffected by terrain rise; and (3) the population was densely clustered along the valley floor.[19]

Modeling receptors located in areas with substantial terrain rise could, however, be a major limitation for exposure modeling systems such as HEM, SCREAM, and GAMS, adversely affecting model results. The automated feature of these exposure modeling systems to estimate concentrations without consideration of terrain rise can lead to large model inaccuracies for source categories, such as power plants and incinerators, that release from high stacks. A bias to underestimate impacts for tall stacks can occur on this basis for areas with moderate to high terrain.

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[19] A limited special modeling analysis was done for Kanawha Valley to demonstrate the limited sensitivity of model results to terrain rise for this application, based on the LONGZ model.

4.6 Meteorological Data

In the ideal case, at least one year of hourly meteorological data collected at a representative location within a study area would be available to support air toxics modeling. The following parameters would be optimal for most of the models used in these studies:

- Wind speed/wind direction
- Atmospheric stability
- Mixing height specific to the study area
- Ambient temperature

In the studies reviewed, three different approaches were used to obtain the required meteorological data: (1) use of available data from one existing site to represent the general study area, (2) use of available meteorological data from multiple sites, and (3) collection of site-specific meteorological data at one or more sites within the study area.

Studies That Used Existing Data from a Single Site—Most studies used available meteorological data from one site in the region to represent a study area. These included the Southeast Chicago study, the NESHAPS study (within the Six Months Study), the Santa Clara study, the 35 County Study, and the 5 City Controllability study. Since all of these emphasized estimation of areawide average concentrations, not MEIs, over annual averaging periods, reasonably representative annual meteorological data from a single site could be expected to satisfy the modeling objectives.

Study That Used Meteorological Data from Multiple Sites with Available Data—The South Coast Study is the only one to use data from multiple existing sites within one defined study area. Here, the closest meteorological station to each source was selected from among 16 available sites. The incremental contributions from each source were modeled on this basis. This procedure was used to account for the differences in wind flow across the relatively large study region, which included substantial terrain complexities in some subareas.

Studies Collecting Site-Specific Meteorological Data—The collection of site-specific meteorological data was only done in these studies to (1) support the interpretation of detailed air quality monitoring programs, as in Philadelphia and Baltimore, or (2) support modeling within a complex topographical setting, i.e., Kanawha Valley.

The IEMP studies in Philadelphia, Baltimore, and Kanawha Valley collected meteorological data at one, two, and four monitoring sites, respectively. Wind speed, wind direction, and turbulence intensity data were collected, generally at 20 m above ground level, at sites selected to best represent the entire study area or subregions within the study area.

In the Philadelphia and Baltimore studies, site-specific meteorological monitoring was primarily done to support the interpretation of the 10-station air quality measured data sets that were collected in each city. For the Kanawha Valley study, meteorological monitoring at three new sites and one existing site was conducted because of the lack of available data to account for the variability in wind flow and dispersion characteristics across four delineated zones within this complex study area.

Treatment of Meteorological Parameters

Differences in the treatment of meteorological conditions can make large differences in modeled concentrations:

Wind Flow—Many of these study areas were relatively flat, such as Philadelphia, Baltimore, and Southeast Chicago, where during most periods the differences in wind flow across the study area would be expected to be relatively small. A likely exception is during evening hours under inversion conditions. During such periods, wind flow can be highly variable and dispersion rates very small. Meteorological data that best represent receptors of greatest concern, such as the locations of air quality monitoring stations, areas with high population density, or MEI locations, are highly desirable for representing such conditions. This can be especially important when specific days are being modeled, such as to support a model performance evaluation.

Stability—Atmospheric stability is a term used within a dispersion model to indicate the rate of horizontal and vertical pollutant mixing within a plume. Unstable conditions produce vigorous mixing, neutral conditions indicate moderate mixing, and stable conditions result in very limited mixing.

Stability can be estimated based on "default" data sources, such as available National Weather Service data, or site-specific data, such as turbulent intensity data. For many applications, default treatments likely provide reasonable estimates and are the only means of estimating concentration because more site-specific data are unavailable. There are applications, however, where default treatments may introduce considerable uncertainty into the modeling analysis.

Research has shown that large differences in dispersion conditions can occur within specific default stability classes (Luna and Church, 1972). The use of default stability data to represent dispersion rates can therefore introduce substantial inaccuracies into the modeling analyses for some applications. An alternative is to collect site-specific turbulent intensity data specifically for the study, as was done in the IEMP studies of Philadelphia, Baltimore, and Kanawha Valley.[20] Based on their experience, two conclusions can be drawn:

1. The influence of terrain on dispersion rates appeared to be relatively significant in the Kanawha Valley study. The benefit of collecting turbulent intensity data as site-specific indicators of dispersion seems greater in this setting, whether the averaging period be short term or annual average.
2. For areas with flat terrain, the site-specific dispersion data appear to be useful for characterizing conditions for short periods, such as for model

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[20] These studies measured the standard deviation of horizontal and vertical wind speed, which was used in conjunction with mean wind speed to estimate turbulent intensity. These data were then used within LONGZ estimate average dispersion rates within each stability class.

performance evaluations or short-term ambient exposures. For annual averaging periods, the default data appear to be a more reasonable indicator of dispersion.

Mixing Height—Mixing height refers to the vertical limit of turbulent mixing. Because toxic air emissions are generally dominated by low-level sources, which are not as significantly influenced by the mixing height term as elevated sources, none of the studies collected site-specific mixing height data. Regionally available mixing height data were used as a default. For relatively large study areas, however, or for modeling the impacts of outlying major industrial areas on central business district receptors, the mixing height term could significantly affect model estimates.

It is important to note that there can be large differences between regionally collected mixing height data and site-specific data. In fact, mixing heights can be quite variable in time and space within a specific metropolitan area. Differences can be most pronounced during evening or early morning hours during low-level inversions.

Although the collection of site-specific mixing heights can be an expensive addition to a meteorological monitoring program, it can at least theoretically improve the representativeness of modeled air toxics concentrations in urban environments. The lack of a mixing height term in the HEM/SHEAR and SCREAM models is a pertinent example in which model estimates might be improved to some degree by inclusion of a representative treatment of mixing height in the modeling analyses.

Treatment of Meteorological Variability

The incorporation of diurnal or seasonal differences in meteorological conditions can be important when modeling industrial facilities or area source categories that have distinct diurnal or seasonal patterns in emission rates. A number of studies took such differences into account.

Diurnal Differences—For modeling purposes, separate daytime and nighttime model input files can be created, containing meteorological and emissions data representative of each period. This approach was used in the Philadelphia, Baltimore, Kanawha Valley, and South Coast MATES studies to minimize diurnal bias, although the results were reported without displaying diurnal differences.

Hours of operation data within the National Emissions Data System (NEDS) are a default data source to aid in apportioning emissions on a day/night basis. More detailed and accurate data on the hours of operation of processes and facilities may be available at the local level. Modeling daytime and nighttime conditions essentially doubles data processing and requires additional quality control, but ignoring these differences can introduce substantial bias in some situations, tending to overestimate concentrations of near-ground-level releases, and underestimating sources with high-level releases. Routine model runs with HEM or GAMS do not consider diurnal differences in release.

Seasonal Differences—Some emission source categories, such as wood burning, have distinct seasonal patterns in their emission rates. Some less obvious source categories can also show distinct seasonal patterns. For example, mobile source emissions are highly sensitive to ambient temperature; winter periods show higher emissions per vehicle mile traveled because of reduced combustion efficiencies (Black, 1986).

If there are also large seasonal differences in transport and dispersion within a study area, bias can be introduced to the modeling analyses if annual average emissions and annual meteorological data sets are used for modeling. This bias could be reduced by modeling on a seasonal basis and by linking seasonal emission rates with meteorological data representative of each season. Annual average concentrations can then be averaged across all seasons. None of the studies used this level of resolution in the modeling analyses, a limitation that most significantly affects heating-related emissions (such as wood and oil burning) and major industrial facilities with emission rates that are highly variable on a seasonal basis.

4.7 Decay, Deposition, and Transformation

The models used in all of these studies contain simplified treatments for decay, deposition, and transformation. These factors generally have not been significant issues in most studies of urban air toxics, but a number of issues deserve at least some consideration.

Decay

The half life values of most pollutants are long in relation to the travel times for traversing the 50 km distances that are the limit of most of these modeling analyses. Over these distances, the exponential decay term contained in these dispersion models would generally produce only minor differences in model output. A widely used, practical simplification in these studies was to assume an essentially unlimited half life, such that the decay term becomes zero and incremental concentrations from each source can be scaled by emission rates for different pollutants.

Deposition

Deposition of pollutants removes pollutants from the ambient air and therefore lowers ambient concentrations.[21] It can occur through three mechanisms: (1) gravitational settling, (2) dry deposition, and (3) wet deposition. Consideration of these terms has not been included in the modeling analyses under review in this study. Model-predicted concentrations may therefore be biased upward on this basis, but in many cases these terms would not substantially affect the results.

Gravitational Settling—Gravitational settling has a significant effect only on pollutants that are bound or attached to particles that are relatively large, such as 10 microns in diameter or larger. Since most directly emitted, particulate phase toxic air pollutants are released from combustion sources with relatively fine particles, little accuracy is likely to be lost by assuming zero gravitational settling for these pollutants.

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[21] Indirect effects of deposited pollutants, such as contamination of surface or ground water, uptake through the food chain, or direct ingestion (pica), have not been considered in any of the current studies and are therefore not discussed here.

The adsorption in the atmosphere of gas phase pollutants onto particulates is another mechanism where gravitational settling could potentially be an issue, but such model treatments are beyond the current state of the art. In addition, a major percentage of the particles involved in the adsorption process likely are less than 10 microns in diameter, and therefore have minimal settling velocities.

Wet and Dry Deposition—Most of the pollutants emphasized in urban air toxics studies to date are volatile organics that are minimally affected by wet or dry deposition because the washout ratios and deposition velocities are relatively low. All of these studies used modeling techniques that assumed that ambient concentrations are not affected by wet or dry deposition.[22] This assumption probably results in slightly higher estimates of concentration than would be predicted if these terms were included.

The wet and dry deposition terms can produce more significant reductions in average concentrations at the urban scale for metals and high molecular weight organic compounds. The current model treatments could have a bias to overestimate concentrations for metals and high molecular weight organics relative to volatile organics.

4.8 Model Execution

For criteria pollutant modeling analyses, the specific procedures in the *Guideline on Air Quality Models* (Revised) (EPA, 1986) are followed. This reference guides model selection, as well as the manner in which the models are to be executed. For example, guidance is provided regarding how to select model options for specific applications. Most of these studies of noncriteria pollutants, however, did not use procedures recommended in the guideline. Alternative model execution procedures were used, apparently because the goals of most studies were perceived to be substantially different from those of typical criteria pollutant modeling programs. The three exceptions (i.e., where the modeling analyses were consistent with the guideline) are Baltimore, Kanawha Valley, and Southeast Chicago.[23]

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[22] The one exception is the version of GAMS used for the 35 County Study. At that time the Atmospheric Transport Model (ATM) was used as the dispersion model in GAMS, rather than ISCLT. ATM could model gravitational settling, dry deposition, and wet deposition. There were substantial limitations, however, noted with the use of these terms (Sullivan and Hlinka, 1986). The wet deposition term was set to zero for the 35 County Study, and the dry deposition term was set to its minimum value. Still, concentrations were probably reduced on the basis of deposition in the 35 County study to a greater extent than appropriate for most toxic air pollutants evaluated in that study.
Another problem that was identified with ATM is an error in the plume rise term (Sullivan and Hlinka, 1986). For sources with relatively high exhaust temperature, such as incinerators or power plants, the 35 County Study would underestimate concentration and risk on this basis.

In terms of the mechanics of executing the model runs, there were two primary techniques used to execute the models: (1) one model run is made for each pollutant based on using actual emission rates and no post processing,[24] or (2) model outputs were made for each source based on a normalized emission rate, such as 1 kkg/yr or 1 g/sec, with post-processing used to sum concentrations across all sources at each receptor.

With the exception of the early analyses in the Philadelphia Study and the NESHAPS Study, normalized modeling techniques have been used in all urban scale toxic air studies, for a variety of reasons. The primary consideration is the relatively large number of pollutants modeled in air toxics studies; for criteria pollutant analyses, where the number of pollutants is much more limited and multiple averaging periods frequently need to be addressed, it is often a more common practice to make pollutant-specific model runs. Another important factor that leads to normalized modeling is the often dynamic nature of emissions inventories that support analyses of the urban soup. Since normalized modeling techniques link modeled data to emissions during post-processing rather than within the modeling program,[25] effects of changes in emissions terms on predicted concentrations can be displayed readily. The same applies to displaying the benefits of various control scenarios. Hence, changes in exposures and risks can be linearly predicted from changes in emissions without having to rerun the models for each emission scenario.

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[23] The Baltimore modeling was consistent with EPA guidance, with the exception that Urban Mode 1 dispersion coefficients were used in place of Urban Mode 3 coefficients. This departure from recommended practices was made because of the study managers' concern that the Urban Mode 3 dispersion coefficients would introduce a bias, underestimating concentrations for the predominant, low-level heights of release.

[24] Post processing involves multiplying normalized concentrations for each source times emissions rates for each source, and then summing, at each receptor, the incremental concentrations across all sources.

[25] Although some dispersion models, such as ISCLT allow for changing modeled data for selected sources, the incorporation of updates and control scenarios in a data base is much easier to implement.

Although the normalized modeling approach is now almost universally adopted, it introduces several significant technical limitations that deserve attention:

- Potential Complications with Atmospheric Decay—Normalized modeling techniques restrict consideration of pollutant decay because differences in the treatment of decay for different pollutants results in the relation between emissions and concentrations for each source becoming nonlinear. As noted above, this is currently not considered an important problem, but there may be situations where decay must be addressed. Although special normalized model runs can be made for pollutants with particularly short half lives, such as the pollutant-specific modeling of allyl chloride in the Kanawha Valley Study, this level of detail can be impractical for a study with wide pollutant coverage.
- Representativeness of Release Specifications—When estimates of ambient concentrations are made for a source on a normalized basis, a set of release specifications are used, such as stack specifications, area source dimensions, and so forth. Some changes in emissions may be associated with changes in release specifications, defying the assumption of linearity inherent in the normalized modeling approach.

4.9 Model Performance Evaluation

Model performance evaluations are only an option for studies that have an adequate measured data set against which to compare the modeled values. Where they are possible, performance evaluations offer two major benefits: (1) they can help evaluate the uncertainties in modeling results—a prime concern when using modeled air toxics data to support regulatory or policy developments, and (2) they can be used as a basis to improve future model performance.

Evaluating Model Performance

Procedures have been established to help support the standardization of model performance evaluations; these include consideration of bias, noise, correlation, and other statistical tests (Fox, 1981). The Philadelphia study showed statistical analyses for each of these tests. The more limited model performance evaluations presented for the South Coast MATES and Kanawha Valley studies emphasized tests of model bias.

Improving model performance implies more than simply comparing statistical analyses of alternative model formulations.[26] Much can be learned about the strengths and weaknesses of alternative model formulations through detailed review of the patterns revealed by the raw measured data set. The measured data can provide clues for improving model performance, such as:

- If measured concentrations decrease with distance from a source inconsistently with the rate of decrease predicted by the model, perhaps the treatment of dispersion is suspect.
- If wide variations are observed in measured concentrations for similar meteorological conditions, perhaps more specific emissions data are needed for key facilities to better describe emissions variability.

For the Philadelphia, Kanawha Valley, and South Coast MATES studies, it appears that the demonstration of model performance strengthened confidence in using modeled data for the exposure assessments. Particularly for studies with complex emissions or topographical factors, model performance testing can help show the reasonableness of predicted concentrations to meet project goals, or demonstrate the failure of modeling techniques to meet goals for some pollutants. Kanawha Valley is a good example of complicated sources located in complicated terrain. Model performance testing in this study, although relatively crude, was used to confirm that the modeled data were at least the correct order of magnitude.[27]

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[26] Formulation refers to a dispersion model run with specific input data and control settings. A dispersion model can be run with multiple formulations.

[27] For this study, the EPA Science Advisory Board (SAB) noted during the series of review meetings the benefits of evaluating model performance in this complex setting. This component of the study appeared to be instrumental in obtaining SAB acceptance of the modeling protocol developed.

Improving Model Performance

It is a common misconception of reviewers of studies that have undertaken model performance testing that measured data are used to calibrate model output to reduce bias, thereby showing a better match with observed results. This has not been done in any of these studies, principally because of the difficulty of demonstrating that model performance on an annual average basis is improved throughout a study area, and not just for the monitoring sites and specific days of the sampling program.

Instead, model performance evaluation is usually used to help reveal the limitations in the physical aspects of a model, such as the strengths and weaknesses of emissions for specific point sources or source categories, weaknesses of treatments of transport and dispersion, and so forth. For example, the Philadelphia study used model performance testing as a basis for developing a priority ranking for independently verifying emissions data. Its procedures and results illustrate many of the issues of concern with model performance evaluation:

Philadelphia Model Performance Evaluation—Four model formulations were tested in the Philadelphia Study, all of which were based on the SHORTZ dispersion model.[28] Different treatments of stability and different methods of considering emissions variability were tested.

Table 4-3 summarizes the means and correlation coefficients across the ten pollutants and ten sites that were included in this program. Figures 4-4 and 4-5 display examples of average modeled and measured concentrations for each of the 31 days of the monitoring program at one of the sites (Site 7) of the Philadelphia network: pollutant coverage is for 1,2 dichloroethane and carbon tetrachloride, respectively.

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[28] As already noted, the Philadelphia study used SHORTZ purely for model performance evaluations, not for evaluating complex terrain. In the opinion of the authors, who participated in the Philadelphia study, it would have been desirable to have tested alternative models, such as CDM and RAM, in addition to SHORTZ.

Table 4-3. Summary of Means and Correlation Coefficients for Modeling and Measured VOCs in the Philadelphia Study

Compound	Measured	Modeled	R Value
Chloroform	0.3	0.2	0.27
Dichloroethane	0.4	0.4	0.89
Carbon Tetrachloride	1.8	0.1	0.47
Trichloroethylene	1.6	1.0	0.00
Benzene	6.0	2.3	0.58
Dichloropropane	1.2	0.5	0.95
Toluene	12.0	8.8	0.76
Perchloroethylene	4.7	3.5	0.88
Ethyl Benzene	4.7	0.4	0.64
Xylene	12.3	2.4	0.23

The IEMP model performance evaluations were performed based on the Monitoring and Ambient Data Assessment Module (MADAM)[29] of the PIPQUIC data management system. Figure 4-6 presents a summary of the inputs and outputs to this system. The Philadelphia study focused on partitioning the measured and modeled data by wind flow quadrant as a means of identifying systematic differences associated with flow across source regions. This approach provided a means of improving the verification of emissions data. The following factors reduced bias, but did not significantly improve correlations.

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[29] MADAM is a software package within PIPQUIC that inputs measured concentrations, normalized modeled concentrations, emissions data, and meteorological data that is concurrent with work days of the monitoring program. Model performance can be displayed through MADAM by partitioning the data by wind flow, stability, or other parameters. See Chapter 7 for a discussion of PIPQUIC.

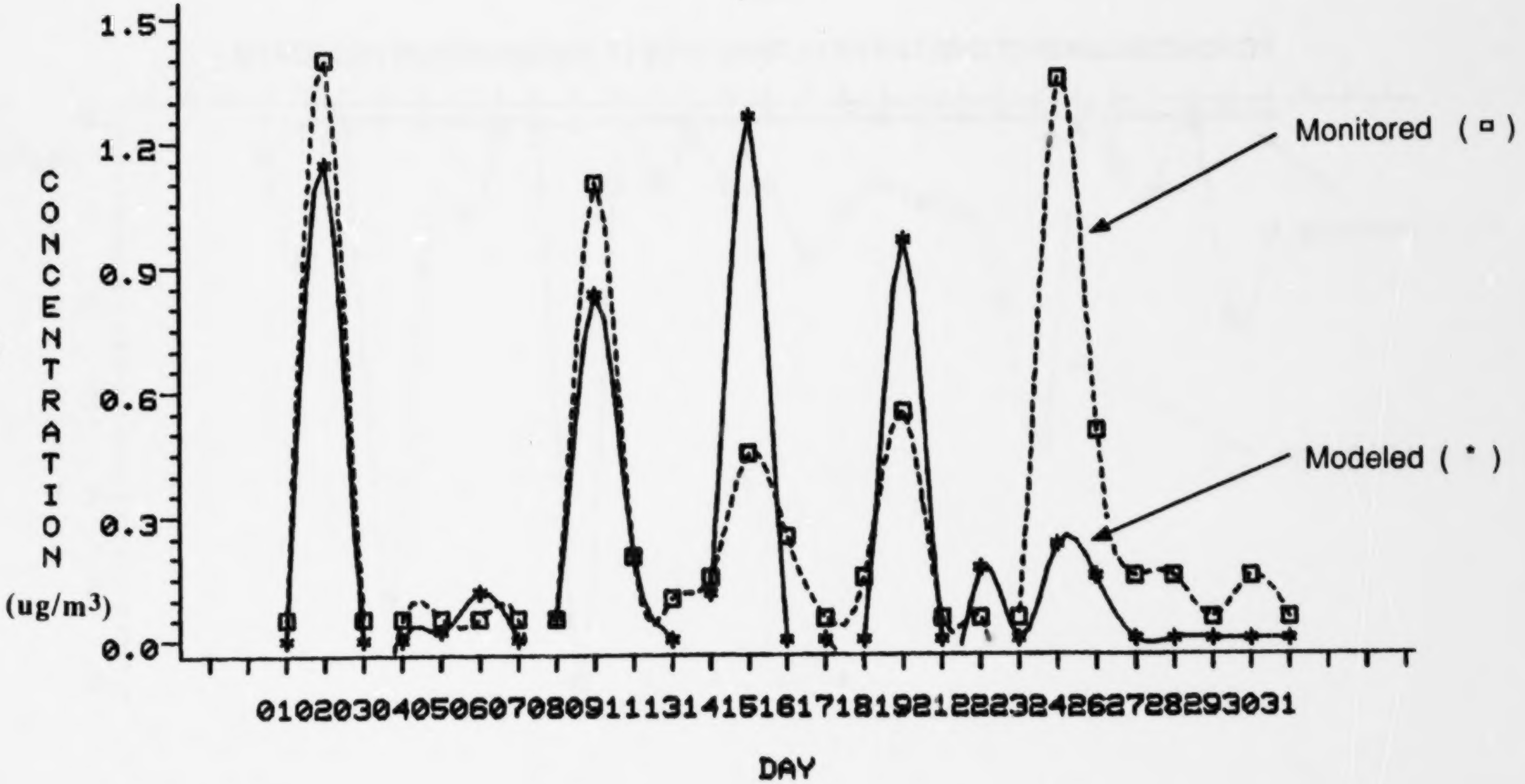

Figure 4-4. Day-by-day comparisons of measured and modeled concentrations of 1, 2 dichloroethane for a site in the Philadelphia monitoring network.

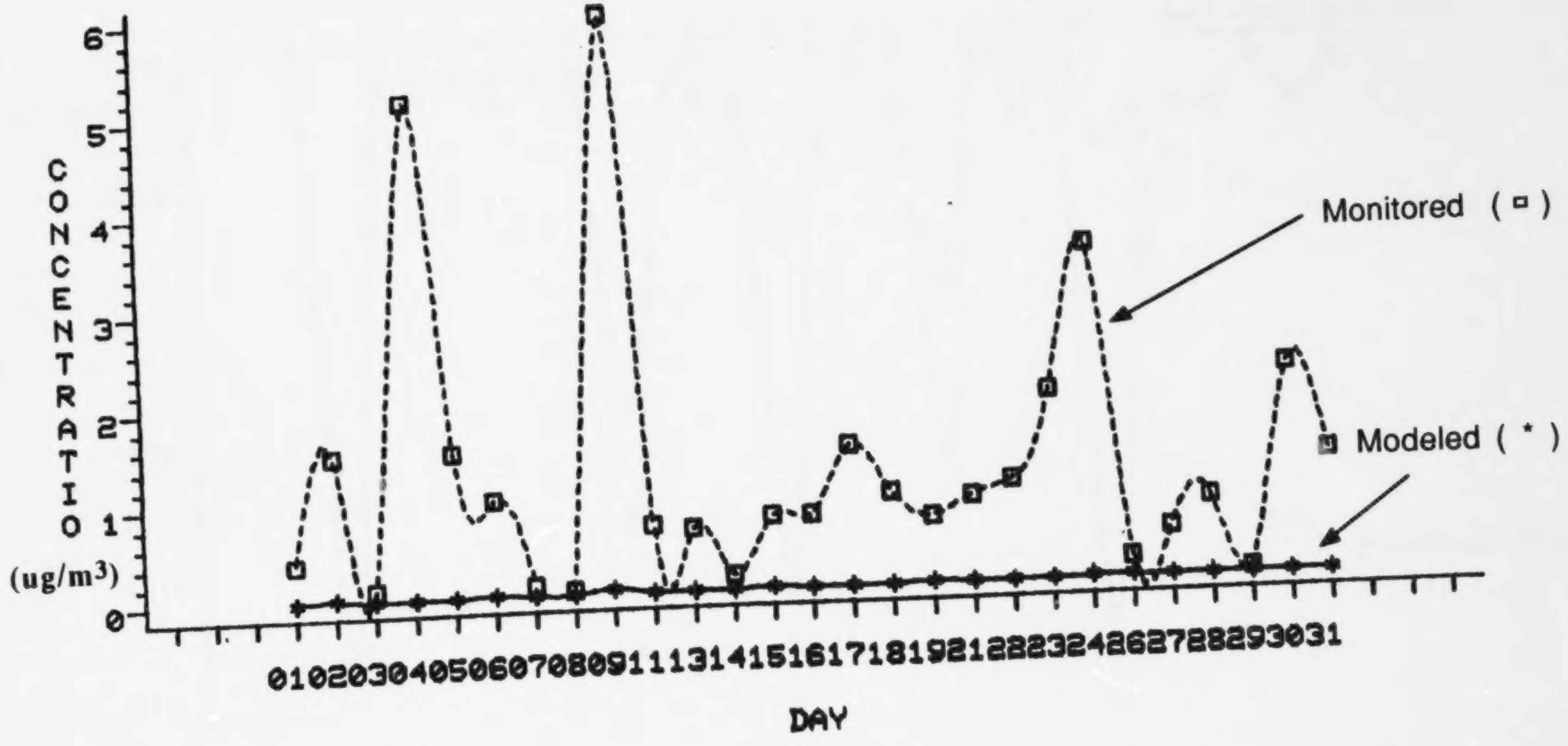

Figure 4-5. Day-by-day comparisons of measured and modeled concentrations of carbon tetrachloride for a site in the Philadelphia monitoring network.

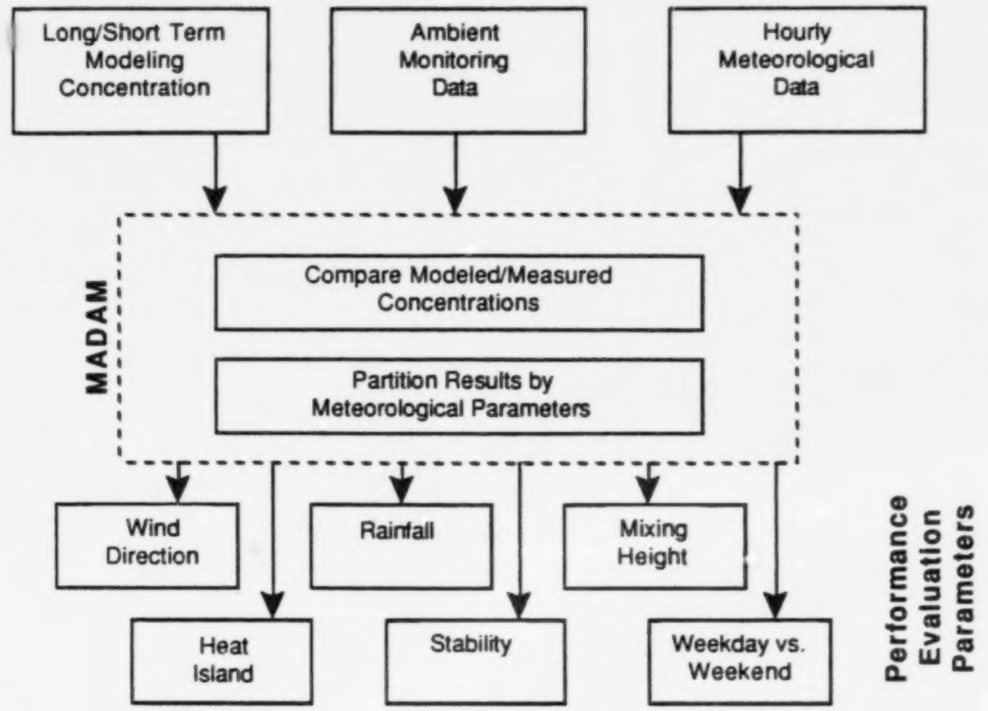
Long/Short Term Modeling Concentration
Ambient Monitoring Data
Hourly Meteorological Data
MADAM
Compare Modeled/Measured Concentrations
Partition Results by Meteorological Parameters
Wind Direction
Rainfall
Mixing Height
Heat Island
Stability
Weekday vs. Weekend
Performance Evaluation Parameters

Figure 4-6. Summary of MADAM Model Performance Evaluation Tool

- The study identified an order of magnitude error in the emission rate for a major facility on this basis. Engineering review by an independent agency identified a process that was missing from the inventory—although process emissions as a category were included in the inventory, the solvent recovery system, which was the dominant source, was inadvertently omitted (Sullivan, 1985).

- Volatilization from industrial sewage en route to the wastewater treatment plant was flagged during the model performance review because of the relatively low concentrations for two key pollutants transferred from an industrial facility to the wastewater treatment plant via the sewage system.

- An area source term was also added for drinking water releases of chloroform, essentially removing the bias for this compound. Again, this

> assessment was done by an independent analyst based on water consumption rates and average chloroform concentrations in the drinking water, not by model calibration techniques.

Only 31 days of monitoring data were available, however, which was an insufficient sample to allow partitioning of the data set in greater detail. It was not possible, for instance, to evaluate performance as a function of mixing height, stability, or similar factors. More data rich monitoring programs, such as the Staten Island study, could provide a more robust data set and permit more comprehensive model performance evaluations for a broader range of toxic air pollutants.

Results of Model Performance Evaluations for South Coast—Table 4-4 presents the limited results of the model performance evaluation in the South Coast area, and Table 4-5 presents the results for the Kanawha Valley. These tables show the ranges of measured and modeled data to provide an indication of the magnitude of model bias. Note that benzene and chromium, which account for about 95 percent of the risk in the South Coast Air Basin, show relatively good agreement between measured and modeled concentrations.

Table 4-4. Comparison of Measured and Modeled Predicted Toxic Air Pollutants in the South Coast MATES Project*

Air Toxics	Measured	Model predicted	Predicted/ measured ratio
ORGANIC GASES (ppb)			
Benzene	1.0–4.9	0.56–5.0	0.22–1.8
Carbon Tetrachloride	0.10–0.12	1.1–24 x 10^{-5}	1.0-25 x 10^{-4}
Chloroform	0.02–0.30	2.0–17 x 10^{-8}	0.68–49 x 10^{-7}
Ethylene Dibromide	0–100	1.1–22 x 10^{-4}	0.11–110 x 10^{-5}
Ethylene Dichloride	0–18	1.7–10 x 10^{-3}	1.2–52 x 10^{-4}
Perchlorethylene	0.5–3.1	0.28–2.4	0.22–1.5
Toluene	2.5–6.7	0.80–3.7	0.16–0.66
Trichloroethylene	1.1–7.1	0.33–2.9	0.13–54
Vinyl Chloride	0–2.0	5.1 x 10^{-5}	2.6 x 10^{-5}
TRACE METALS (ng/m^3)			
Arsenic	0–8.8	5.0–10 x 10^{-4}	1.5–2.3 x 10^{-4}
Beryllium	0–0.5	0–5.4 x 10^{-3}	0.003–3.4
Cadmium	0–4.1	1.1–9.6	0.71–1,200
Chromium	1.8–11	3.6–60	1.06–8.6
Lead	180–280	1,100–1,700	3.9–9.4
Nickel	3.7–8.9	0.7–5.6	0.08–7.3

*This table shows the ranges in measured and modeled concentrations across all of the "existing" sampling sites, which are more indicative of average/background concentrations than the SCAQMD's "new" sites, which were selected to reflect maximum exposures.

Table 4-5. Comparison of Measured and Modeled Data for Kanawha Valley Toxics Screening Study

Concentration (µg/m^3)

	Site 1		Site 2		Site 3		Site 4	
Pollutant	Mod.	Mon.	Mod.	Mon.	Mod.	Mon.	Mod.	Mon.
Chloroform	1.9	3.6	4.5	8.7	8.1	8.9	11.5	8.0
Methylene Chloride	0.6	3.3	1.3	3.1	10.8	20.8	14.6	12.3
"Ethylene Oxide" (unknown compound)	2.4	8.4	12.8	11.3	0.0	13.3	0.0	12.4
Carbon Tetrachloride	0.0	1.4	0.0	1.0	2.6	3.4	3.3	2.2

NOTE: Ethylene oxide ("unknown compound") averages are based on very limited data: Site 1 (n=10); Site 2 (n=14); Site 3 (n=3); and Site 4 (n=4).

Conclusions Drawn from Available Model Performance Evaluations to Date—The limited model performance evaluations completed to date in these three study areas tend to support the following general conclusions:

- Models tend to underestimate urban concentrations of many toxic air pollutants. It is obvious from review of Tables 4-3, 4-4, and 4-5 that, in general, there is a bias to underestimate modeled concentrations relative to ambient concentrations for volatile organics. (From Table 4-4, no clear trend exists for metals.) Similar model underestimations for volatile organics were found in other studies (EPA, 1988; Sullivan and Martini, 1987). In many cases modeled results are low by a factor of 2 or 3 or more. Four factors may be responsible for this tendency toward underestimation:
    1. All of the model-based analyses assumed zero background. For some of these compounds, especially those with long atmospheric half lives such as carbon tetrachloride, the background term is probably a substantial portion of total concentrations.
    2. Gaps in coverage in emissions inventories could be accounting for a significant percentage of the model underestimates.
    3. Current models do not predict secondary pollutant formation (e.g., photochemical formaldehyde).
    4. For the more recent studies (EPA, 1988 and Sullivan and Martini, 1987) that evaluated model performance based on the EPA recommended dispersion coefficients for urban applications (i.e. Urban Model 3 dispersion coefficients), substantial bias to underestimate concentrations was observed for low-level sources relative to the more traditional Pasquill-Gifford coefficients that are modified for urban applications.

- Model performance appears to be strongly pollutant dependent. These studies appeared to be successful in separating pollutants with relatively low bias (such as less than a factor of 2) from those with high bias, which in some cases was orders of magnitude. These findings help support the use of modeling for exposure assessments for the subset of pollutants found to be within reasonable accuracy limits.[30]

The correlation of the average concentrations across all sites is another important factor with which to judge model performance, particularly to evaluate the ability of a model formulation to identify gradients in concentration across a study area. The Philadelphia study is the only study to report correlation results; its correlation coefficients (R), presented in Table 4-3, indicate wide differences in correlation across pollutants, ranging from 0 to 0.95. When considered in conjunction with bias, as shown in Table 4-4, it is clear that far more confidence should be placed in the results for some compounds than for others. For example, modeling predictions of carbon tetrachloride or arsenic levels appear to be orders of magnitude less reliable than pollutants that appear to be better characterized in emission inventories, such as perchloroethylene or toluene.

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[30] In many cases there also is considerable uncertainty in the measured data, which should be considered when assessing the confidence in the modeled data.

4.10 Insights into the Use of Modeling in Air Toxics Evaluations

The experience gained through the air toxics studies conducted to date lead to insights that can guide future use of dispersion models in multi-facility, multi-pollutant studies. Observations on their experience are grouped below under four headings: (1) Modeling Techniques, (2) Model Performance Considerations, (3) Completeness of Scope, and (4) Cost Saving Measures.

Modeling Techniques

There are many similarities between routine modeling analyses done for permitting or State Implementation Plan (SIP) development for criteria pollutants, on the one hand, and modeling done to support multi-pollutant, multi-facility urban air toxics studies, on the other. In many cases the same Gaussian dispersion models are applied, with similar meteorological and release specifications data. There are also some major differences. The following are issues that appear to be of greatest concern for modeling noncriteria pollutants:

Background—All of these modeling analyses were limited to pollutants released within the study area. There was no inclusion of a background term to estimate concentrations in the ambient air (before transport into the study area) prior to the incremental contribution from the pollutants released in the study area. This background term is not necessarily represented by available data from remote rural sites, but should be representative of concentrations at the upwind fringes of the modeling domain. Although in some cases, such as the Philadelphia and Southeast Chicago studies, the boundary of the modeling domain was substantially larger than the receptor area to partially account for local transport, any regional background was not effectively considered. Available national data bases, which cover a wide range of monitoring sites, suggest that some of the pollutants commonly reviewed may have background levels that are significant in relation to modeled concentrations (Sullivan and Martini, 1987). The increased ambient monitoring by EPA and others should improve information on pollutant background levels, at least in the urban context. Little rural sampling is under way; if attempted, it would probably yield nondetectable levels for some pollutants.

Tendency to Underestimate Modeled Concentrations for Some Compounds—Although limited data are available, the results in Table 4-4 suggest that model underestimates can be relatively large for pollutants such as carbon tetrachloride, chloroform, ethylene dibromide, xylene, and ethyl benzene. These and other pollutants suspected of being substantially underestimated by modeling should be (1) pursued to evaluate cause or (2) flagged in the reported results.

Use of Normalized Modeling Techniques—When normalized model runs are used to support the analysis of control options, there is a potential for mischaracterizing release specifications for alternative controls. As processes change, and especially if controls are added, the release specifications can substantially change, requiring follow-up modeling of these sources to replace outdated normalized model output. For example, if a scrubber is assumed to be added to enhance the control of emissions from a facility, the release specifications can be substantially altered. In this example, the benefits of a control option could be greatly exaggerated because the lower effective release height of the scrubber would not be considered if only the resulting emissions were adjusted in the modeling analysis. (A scrubber can produce a substantially lower exhaust temperature, which can result in much lower plume rise and can increase the air quality impacts on a unit mass release basis.) This issue can be a problem especially for data bases that are used on a long-term basis as a repository for emissions and modeled data. Some system controls are needed to maintain the currency of the data base, including flagging major process changes and changes in facility status through the permit process.

A problem related to the representativeness of the release specifications is maintaining the currency of the emissions data in a data base. If a normalized modeling approach is selected to meet the immediate project goals, and to become part of a data base for future uses, a system is needed to ensure that emission rates and release specifications remain current.

Microscale Errors for Simplified Release Specifications—Most facilities in these studies were modeled with highly simplified release specifications. In some cases this likely was a reasonable approach, especially for estimating average concentrations across a study area. In the case of modeling MEI concentrations for highly complex facilities, however, major inaccuracies likely resulted from using simplified release specifications to estimate maximum offsite annual average concentrations. Depending on the layout of each facility, it is possible that modeled concentrations could underestimate or overestimate on this basis. The study that appears to best represent MEI concentrations is the Kanawha Valley study, which is consistent with the emphasis provided on the modeling goals in Table 4-1.

Modeling Fugitive and Vent Releases—Fugitives and vent releases are dominant source categories for many major and minor industrial releases of toxic air pollutants. In many cases, stack emissions make minor contributions, yet the emphasis on model development within EPA has been on releases of criteria pollutants from stacks. Errors within area and building source treatments within models such as ISCLT and LONGZ can introduce bias that acts to underestimate the risks from air toxics (Sullivan and Hlinka, 1985). These two models were found to have an error in the smoothing term that acts to underestimate concentration from area or building source treatments. It is unclear, based on Sullivan and Hlinka, 1986, if this error is present in other models, such as CDM.

Coupling modeling uncertainties for fugitive/vent releases with the often large uncertainties in emission terms adversely affects the confidence in modeling many major point sources of air toxics. Model performance testing within the composite plume (such as 1 km from the fenceline of a facility, as used in Kanawha Valley) provides a means of displaying the overall acceptability of the modeling analyses for such sources.

Model Bias for Low-Level Sources in Urban Areas—It does not appear that any of these modeling analyses are based on the most recent set of dispersion coefficients recommended for modeling in urban areas. This is largely because most of these studies predate this guidance (EPA, 1986). The UNAMAP 6 version of the EPA dispersion models provides regulatory guidance for using an adaptation of the McElroy-Pooler dispersion coefficients for urban applications. These coefficients predict concentrations from low-level sources that are substantially lower than predicted by previous versions of the UNAMAP models, which may result in significant underestimations of the magnitude of air toxics risks (Sullivan, 1985). It is also likely that source culpability will be inaccurately apportioned between low-level and high-level releases (such as power plant or incineration emissions) and low-level industrial and area sources. (EPA, 1987)

Transformation Products—Limited laboratory testing has indicated that the reaction products of some pollutants have order of magnitude higher mutagenicity than the parent compounds. This is a research issue that is beyond the scope of applied modeling studies conducted at this time, and likely in the near future. Some transformation products (e.g., secondary formaldehyde) may be covered by aldehyde monitoring, but others likely are missed by both modeling and monitoring components of current multi-pollutant, multi-facility studies. Although little can be done to fill this void in present studies, it is an issue that should be considered when interpreting the magnitude of the overall results.

Short-Term Variability in Concentrations—There has been no comprehensive treatment of the effect of the variability of emissions and meteorological data on the short-term variability in modeled concentrations in an urban area, or in the immediate vicinity of major industrial sources. For air toxics evaluations, the issue of most likely potential future concern would be noncarcinogenic health effects related to maximum short-term concentrations. Studies based on annual average emissions rates and the use of stability class data could underestimate peak concentrations by orders of magnitude (Luna and Church, 1972).

To date, it does not appear that any studies have adequately screened the magnitude of potential for exceedance of short-term noncarcinogenic thresholds, either region-wide or within isolated subregions. From a modeling perspective, the issue is accurate representation of dispersion rates on a short-term (hourly) basis and estimation of hourly distributions of emission rates for key selected industrial processes (and possibly area source categories).

Two EPA efforts currently in the planning stages may contribute to the understanding of short-term variations in concentrations of air toxics. EPA's Office of Research and Development is initiating a study to characterize variations in concentrations of a wide range of toxic air pollutants in urban areas. In addition, a follow-up study is planned in the Kanawha Valley to address the distribution of concentrations of selected toxic air pollutants. These studies could help show the relationship of peak hourly concentrations to data compiled to estimate thresholds for toxic air pollutants.

Absence of a Mixing Height Term in HEM/SHEAR and SCREAM—The mixing height term can result in significantly higher predicted concentrations when modeling at the scale of a metropolitan area, or when assessing exposure for major industrial facilities that are located 10 to 15 km or more from high population centers. The lack of a mixing height term in SHEAR and SCREAM appears to be a simplification that could underestimate some concentrations and risks.

Use of Turbulence Intensity Data to Characterize Dispersion Rates—The Kanawha Valley study demonstrated the potential benefits that could be achieved if a dispersion model, such as SHORTZ or LONGZ, were modified to include functional relationships between turbulence intensity data and site-specific dispersion rates representative of near ground-level releases.

Availability of Terrain Data—The HEM and GAMS exposure modeling systems could be refined to include access to nationally available terrain data. This could lead to more accurate model prediction, especially for tall stacks located in moderate to rough terrain. Without consideration for terrain, the model estimates of power plants, incinerators, and other sources with high effective stack heights may underestimate certain ambient concentrations.

Network Size for Model Performance Evaluations—A common question posed during the design stages of a study involves the optimal number of monitoring sites and the length of the field program that is needed to support a model performance evaluation.

The duration of most of these monitoring studies was one season, with intermittent sampling. As long as meteorological and emissions data are available to compare this season to annual and climatological conditions, the single season approach appears to be a reasonable tradeoff between cost and data needs. There also is precedence for single season model performance testing for criteria pollutant applications.

Appendix B presents a methodology to help address the issue of the optimal number of monitoring sites, but ultimately the size of the network needs to be selected to match the goals of the model performance evaluation. For example, if only model bias is evaluated, as presented in the South Coast Study, a few carefully selected sites may meet this objective. If spatial correlation were also to be assessed, such as in the Philadelphia study, it would appear more reasonable to design an eight to ten site network. If costs were prohibitive, perhaps some of the cost-saving measures as presented below could be considered to stretch monitoring resources.

Cost-Saving Measures

Screening versus Refined Modeling—An obvious cost-saving measure for modeling analyses is to first do screening-level analysis prior to performing refined analysis. By using coarse modeling grids and simplified release specifications, it is possible to identify the geographic areas and sources that require the greatest emphasis in the more detailed modeling to follow. In this manner, available resources can be most efficiently allocated to improving estimates in high impact areas.

Normalized Emissions Modeling—Modeling of normalized, rather than actual, emissions is discussed earlier in this chapter. This technique saves resources insofar as it eliminates the need to rerun the model(s) every time emissions change or a different control scenario is evaluated (assuming that the release specification for each facility does not change significantly as emissions change).

Chapter 4

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CHAPTER 5

EXPOSURE AND RISK ASSESSMENT

The following topics will be discussed in this chapter:

- The use of exposure and risk assessment in air toxics studies
- Methodological issues
- Comparison of approaches to interpretation of exposure and risk
- Comparison of field-oriented and research-oriented approaches to exposure and risk assessment
- Insights into the use of exposure and risk assessment in air toxics studies

5.1 The Use of Exposure and Risk Assessment in Air Toxics Studies

Estimating exposures and risks associated with air toxics is a complex issue that most studies have addressed in a highly simplified manner. Total human exposures to air toxics include contributions from numerous residential, commercial, occupational, and transportation-related microenvironments. Fully describing these exposures and their consequent risks is beyond the scope of field-oriented studies concerned with ambient outdoor air concentrations of air toxics. The Total Exposure Assessment Methodology (TEAM) program has attempted more comprehensive exposure evaluations, but has not attempted detailed examinations of individual microenvironments. The Integrated Air Cancer Project, on the other hand, is investigating microenvironments, but is not conducting comprehensive exposure assessments, such as through the personal sampling methods employed by TEAM. Evaluation of all risks associated with air toxics is hampered by gaps in available cancer and noncancer potency data; much additional research must be completed before reasonably comprehensive evaluations of all potential air toxic risk can be attempted.

All of the studies reviewed in this report evaluated cancer risks from air toxics. With the exception of the TEAM studies and the Integrated Air Cancer Project, all the studies simplified their exposure and risk assessment in a similar manner, essentially reducing them to simple multiplications of ambient concentration values times population exposed times available cancer potency scores, as shown in the following formula:

$$\begin{matrix}\text{Areawide}\\ \text{Cancer}\\ \text{Incidence}\end{matrix} = \sum \begin{pmatrix}\text{Model Predicted}\\ \text{or}\\ \text{Measured Ambient}\\ \text{Concentrations}\end{pmatrix} \times \begin{pmatrix}\text{Number of}\\ \text{Persons Exposed}\\ \text{to}\\ \text{Ambient Concentrations}\end{pmatrix} \times \begin{pmatrix}\text{Cancer}\\ \text{Unit}\\ \text{Risk}\\ \text{Factor}\end{pmatrix}$$

In this formula, areawide, excess, additive (i.e., multi-pollutant) cancer incidence is computed by summing over each subarea (e.g., 1 x 1 km^2 grid square) and each pollutant. Cancer unit risk factors are, with some exceptions, drawn from current listings developed by the EPA Cancer Assessment Group and are accepted as constants. These factors are available from EPA's Integrated Risk Information System (IRIS). (EPA, 1988) Some groups used other unit risk factors for certain pollutants. The South Coast MATES project, for example, used risk factors developed by the California Department of Health Services, as required under State law. (Barcikowski, 1988) The issues of greatest practical concern are (1) estimating concentrations (usually by monitoring or modeling) and (2) assigning population to these concentrations. Both topics have been discussed in earlier chapters. This chapter therefore deals primarily with the purposes to which exposure and risk assessment have been put and how their results have been interpreted. Of special interest are studies that compare results obtained through monitoring with those obtained through modeling. It is also important to discuss differences between typical field methods and more refined methods explored by research programs such as IACP or TEAM.

At least two of the studies reviewed—the Baltimore and Santa Clara IEMPs—also evaluated noncancer health effects. In the Baltimore IEMP, the investigators calculated a "health hazard index" to assess the potential hazards from cumulative exposures to the target compounds. The index is based on the assumption of dose additivity and is defined by the following equation:

$$HI = \frac{E_1}{AL_1} + \frac{E_2}{AL_2} + \ldots + \frac{E_i}{AL_i}$$

Where:

HI - Health index for a particular category of health effect
E_i - Ambient concentration of pollutant i
AL_i - Threshold value for pollutant i for a particular category of health effect

After first calculating the ratio of the ambient concentration for the pollutant to the threshold for the health effect (E_i/AL_i), these ratios are then summed across all pollutants with the same effect.

The health hazard index is a numerical indication of the level of concern associated with exposures to complex mixtures of pollutants in the environment. As the index approaches unity, concern for the potential hazard of the chemical mixture increases. If the index exceeds 1, the concern is the same as if the no-effect threshold were exceeded by the same amount by an individual pollutant.

As noted in earlier chapters, the studies under review have varied in the types of exposure and risk estimates they developed. Most, for instance, emphasized areawide or grid cell average annual population exposures; only a subset estimated values for "most exposed individuals" (MEIs). Beyond this, however, the interpretation of results follows some common patterns.

- Ranking the relative importance of pollutants: One of the most common analyses has been to rank pollutants in order of importance, either in terms of ambient concentrations or in terms of potential risk (ambient concentration times potency). Where "threshold" (most noncarcinogenic) pollutants are concerned, the ranking may simply distinguish between pollutants whose concentrations fall above health thresholds and those that do not.

- Ranking the relative importance of sources or source categories: The other most common analysis is to investigate the relative importance of sources or source categories—for single pollutants, for common pollutant groups, or for total risk. Source categories can be broad (area sources versus point sources, mobile sources versus stationary sources) or narrow (discrimination among industrial types). In some cases, total risks or pollutant-specific risks are compared from source to source. Any of these analyses can help set priorities for potential future source controls.

- Estimating "culpability" of particular sources or source groups in regard to individual receptors: Culpability analysis determines, for a particular grid cell or receptor point, the risks or pollutant-specific ambient air concentrations at that point which are attributable to specific sources in the area. This type of analysis may be most useful for MEI analysis, where the immediate question after identifying the most exposed individual is to establish which sources are responsible for that exposure.

- Evaluating variations in exposures and risks: For studies that estimate MEI exposures and risks, the ratio of MEI to average values may be of considerable interest. For all studies, the ranges of exposures or risks are also of interest, particularly in terms of the number of people exposed to various levels. Note that population risks can be characterized in two significantly different ways: (1) as the total number of cancer or other disease cases estimated within the exposed population, a figure that is inherently dependent on the size of the exposed population (some studies based on population normalize their incidence figures to avoid this dependency), and (2) as probabilistic risk (i.e., ranging from 0 to 1) experienced by the average exposed person in the region or in each grid cell, a figure that is not dependent on the total number of people in the region.

- Evaluating relative effectiveness of alternative control scenarios: The 5-City Controllability Study and several of the IEMPs projected future risk reductions as a function of alternative combinations of measures superimposed to control air toxics emissions (see Chapter 6).

5.2 Methodological Issues

Although many of the technical issues involved in the initial steps of exposure and risk assessment have already been discussed in earlier chapters, there are a number of special topics that require additional clarification. This section also briefly summarizes risk assessment methods and available data.

Exposure Assessments

Most technical issues pertinent to conducting exposure assessments have already been discussed in previous chapters. These included such factors as:

- Selection of study boundaries

- Location of monitoring sites in relation to sources and population
- Design of receptor grids and special receptors for modeling analyses
  - —Type (polar, rectangular)
  - —Density
  - —Special Points (monitoring station locations, known MEI locations)

Various other exposure-related topics are discussed below.

Linking Population Data with Concentration Data—The South Coast MATES study, which estimated exposure based on the direct use of measured data, employed the most detailed treatment for assigning concentration data to population of any study reviewed. The study first plotted the locations of the study's ten monitoring stations on its 4,200-cell grid, each cell being 1 km on a side. It then estimated average concentrations for each grid cell by averaging the reported concentrations at each monitoring station, weighted in inverse proportion to the square of the distance to the cell from the monitoring station. In other words, concentrations at a monitoring station 2 km from a particular cell would be given one-fourth the weight of concentrations 1 km from a cell. All population within a grid cell was then assumed to have the same average concentration, based on the distance-weighting procedure. (SCAQMD, 1987)

The 5 City Controllability Study and NESHAPS component of the Six Months Study utilized EPA's Human Exposure Model (HEM) to link population and concentration data. HEM is an integrated modeling-exposure assessment system that first runs a dispersion model to estimate ambient air concentrations around sources and then assigns these concentrations to BG/ED population centroids, based on internally stored 1980 census data. HEM calculates a polar concentration array for 160 receptors around each point source (ten receptors extending radially out to 50 km along each of 16 wind directions). See Figure 4-3 in Chapter 4. Then, depending on whether one is running HEM/SHED or HEM/SHEAR, HEM associates concentrations and populations in different ways as a function of distance from the emission source. Within 3.5 km, the polar grids are smaller than typical BG/EDs, and SHED apportions BG/ED populations to each grid point concentration based on proximity and area. Beyond 3.5 km in SHED, and at all radial distances in SHEAR, a log-log linear interpolation is made among the polar grid points to estimate concentrations at each BG/ED centroid. Incremental contributions from multiple facilities in an area are stored at the BG/ED level in SHEAR and summed across all facilities. Total exposures are then estimated in SHEAR across all BG/EDs in the study area. The SHED methodology is considered more accurate than SHEAR for estimating maximum lifetime risks, but not significantly more accurate for estimating aggregate incidence.

Within HEM, area sources are handled only in the SHEAR module. The user does not provide subcounty-allocated data to SHEAR; rather, SHEAR performs this subcounty allocation in either of two user-specified ways—(1) uniformly, assuming no spatial variation in emissions, or (2) by population. In the latter case, a simplified box model is run within hypothetical boxes superimposed over each BG/ED. The resulting concentrations are therefore associated directly with population controls of each BG/ED.

In the IEMP studies, a rectangular grid was used rather than a polar coordinate system, as is used in HEM. The census tract maps for the metropolitan areas were then overlayed onto the modeling grid and population assigned to each grid cell. For census tracts that extended into two or more grid cells, population was assigned to each affected grid cell on the basis of the percentage of the area of the census tract within each cell. Estimates of exposure for each grid cell were made by multiplying the resulting grid cell populations by the model-predicted concentrations for each grid cell. In several of the IEMPs, up to four receptor points were defined within each grid cell; in these instances, the average concentration from these multiple points was multiplied by the grid cell population to estimate exposure. Total exposure was estimated by summing across all grid cells.

A number of quite different approaches for relating populations and concentrations are planned for the Denver study. As noted earlier, most toxic pollutants in the Denver area are believed to be mobile-source related, with concentration peaks near the urban center; hence, more homogeneous concentrations are expected here than in a more industrialized area. Because of this anticipated regional homogeneity, only three monitoring stations will be used, but, as discussed in Chapter 2, pollutant coverage at each site will be extensive. One goal of the study is to represent uncertainty in concentrations by estimating ranges of exposures as well as best estimates. Plans for defining these ranges include:[1]

1. Calculating expected exposures by simply assigning measured values at each of the three monitoring stations to the population residing in each station's delineated zone.

2. Calculating exposures at distances away from the three monitoring stations through a variety of statistical correlations, one of which is expected to be multiple regression analysis involving (1) available conventional pollutant data and (2) distance from the center of the city. CO data would serve as an indicator of all automotive-related toxics, which include all categories of pollutants of concern in the study (metals, VOCs, semivolatiles, and aldehydes/ketones). Fine particulate data would be correlated with metals concentrations, and ozone data with aldehydes/ketones. Distance from the center of the city is significant because empirical evidence suggests that, despite regional mixing, air pollutant concentrations peak in the central urban area.

3. Estimating high and low average concentrations across all three monitoring sites for each pollutant to define endpoints of ranges of exposures in the region.

Population Mobility/Microenvironmental Exposure—Population mobility has not been widely addressed in studies to date. Of the studies reviewed, only the Motor Vehicle Study attempted to factor in non-outdoor exposures, using a modified version of EPA's NAAQS Exposure Model (NEM) for CO. (Ingalls, 1985) The NEM approach relies on an activity pattern model that simulates a set of population groups called cohorts as they go about their day-to-day activities. Each of these cohorts is assigned to a specific location type during each hour of the day. Each of several specific location types in the urban area is assigned a particular ambient pollutant concentration based on fixed site monitoring data. The model computes the hourly exposures for each cohort and then sums up these values over the desired averaging time to arrive at average population exposure and exposure distributions. Annual averages are possible because a full year's data from fixed site monitors are input to the model.

It should be noted that indoor concentrations (and, therefore, exposure) caused by ambient mobile source pollutants are also accounted for in the model. A scaling factor of 0.85 was applied to the appropriate neighborhood monitoring data to estimate indoor exposures to the pollutant of interest in each neighborhood. The scaling factor was based on comparisons of indoor and outdoor CO levels of homes with no indoor CO sources (e.g., gas stoves, smokers).

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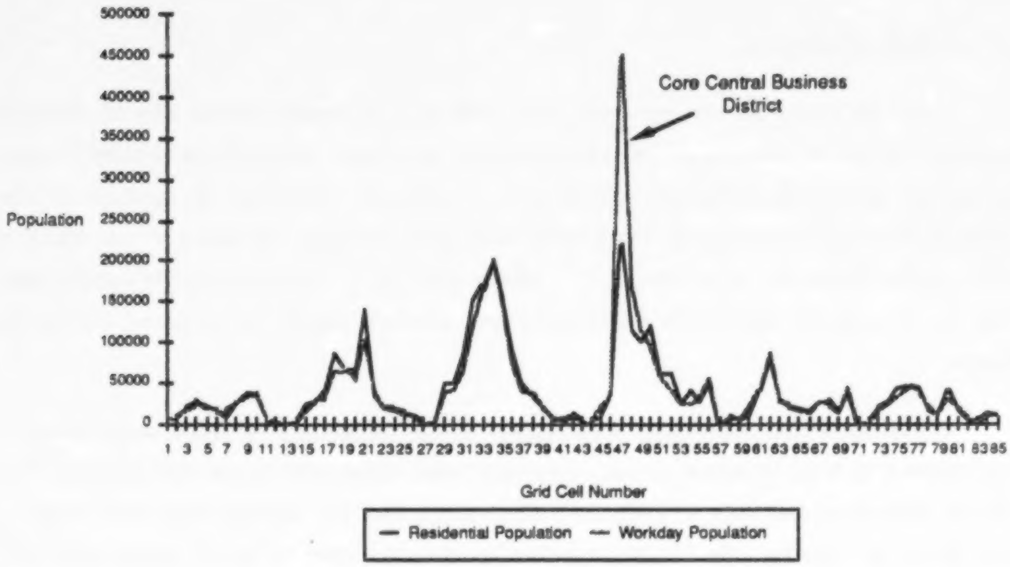
[1] These approaches may not necessarily be carried out as described here.

The modified NEM does not account for photochemical reactions. The exposure levels predicted by the model are those resulting from direct exhaust emissions, and do not account for either the destruction or photochemical formation of the pollutant in the atmosphere. The model also assumes that the pollutant of interest has emission formation and dispersion characteristics similar to those of CO.

In contrast to the NEM approach in the Motor Vehicle Study, most studies, reviewed herein implicitly assume a fixed population defined by the place of residence. To evaluate the reasonableness of this assumption, Figure 5-1 summarizes the results of the work done by Rheingrover (1984), which helps shed light on the assumption of a nonmobile population. This figure applies to the Philadelphia Metropolitan area, as represented by 85 five-km grid cells.

Figure 5-1. Comparison of Residential and Workday Population by Grid Cell



This figure shows population totals (1) under the assumption that all residents remain at their place of residence, and (2) as compared to workday estimates that account for occupational mobility. As shown, most grid cells had relatively equal residential and workday populations, reflecting an equal influx and outflux of people during the work day. The core central business district (see grid cells 47 and 48 in Figure 5-1) shows a noticeable exception, where the influx of workers did substantially increase exposures during the daytime over what would have been estimated based on considering residential exposures only.

If this is representative of other metropolitan areas (which has not been demonstrated to date), the assumption of no mobility for ambient-based exposure assessments does not appear to be a major limitation, at least in terms of outdoor air exposures. A more important factor likely is the mobility of subjects among different microenvironments (e.g., indoors, automobiles, etc.). Only personal sampling and microenvironment sampling are able to account for the mobility of the population beyond outdoor air exposures.

Risk Assessment

The air toxics studies reviewed here used existing health effects data to estimate potential air toxics risks based on their exposure estimates. None of the applied studies generated any original health effects data. Although technical discussion of the development and interpretation of dose-response data is beyond the scope of this study, a brief review of the use of existing health effects data for air toxics studies is appropriate. The use of potency figures for carcinogens and noncarcinogens is discussed separately below.

The factor that contributes significantly to the uncertainty in urban assessments is the conversion from exposure to risk, especially when using unit cancer risk factors. The use of limited animal data to evaluate human risks, and the extrapolation from high to low doses, are two contributors to uncertainties that can span orders of magnitude. For air studies, additional uncertainties also exist when oral ingestion effects data are used to estimate potencies of the same substances when inhaled; in the absence of other evidence, researchers must often make the assumption that oral and inhalation potencies are the same.

Carcinogens—Cancer has been the major emphasis of these studies, with nearly all analyses being focused on this effect. All of the studies that performed exposure/risk assessments relied heavily on unit risk values developed by the EPA Cancer Assessment Group (CAG). These are usually expressed as "unit risk factors," a number that represents the probability of contracting cancer from constant inhalation, over a nominal 70-year lifetime, of 1 $\mu g/m^3$ of the substance in question. These are established conservatively, generally representing the 95th percentile upper bound value based on laboratory experiments.

Studies that employ these dose-response functions explicitly or tacitly assume the following:

- Risks calculated are theoretically valid only for constant exposures over a 70-year period. Some studies have "annualized" cancer incidence or risk values by dividing lifetime risk or the total number of cases by 70. Although this is intuitively useful for policy purposes, it is not, strictly speaking, a valid application of these numbers.

- There is no threshold of exposure below which no carcinogenic effects are presumed to exist. Any dose of a carcinogenic substance, no matter how small, is assumed to impose a finite risk of contracting disease. For most studies of carcinogenicity at the low levels encountered in ambient air, the shape of the dose-response function is assumed to be linear.[2]

- Cancer effects are additive. Studies have assumed that it is valid to sum the risks of different carcinogenic substances. For instance, if substances A and B are each assumed to impose a risk of 10 cancer cases over 70 years to a given population, then the additive risk for the two together is 20 cases.[3]

- No interactions exist among chemicals that would cause simultaneous exposures to multiple carcinogens to lead to either higher or lower disease incidence than would otherwise be presumed to occur. CAG potency values include no allowance for synergistic or antagonistic effects. According to EPA guidance, all cancer incidence estimates are assumed to be additive unless there is specific evidence of interaction among substances. So far, no such evidence is available with reference to the concentrations of carcinogens that may found in the ambient air.

- No cancer occurs from secondary or transformation products or other compounds for which cancer potency scores are unavailable.

Since all dose-response values published by CAG have undergone extensive peer review, individual studies generally have not had to defend the validity of any toxicological potency estimates developed by this group. Some studies, however, such as the Kanawha Valley and Southeast Chicago studies, have augmented the CAG list with additional unit risk values developed by other offices within EPA. For example, in Kanawha Valley other unit risk values were used for formaldehyde and propylene oxide.

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[2] Obviously, risk levels cannot exceed 1, but the shape of dose-response curves as they approach 1 will vary. This issue is assumed to be unimportant in ambient air studies because measured or predicted ambient concentrations are almost always at the very low end of the observable range.

[3] The theoretically more correct approach is to assume that effects should be summed only for carcinogens affecting the same organ. For instance, risks from all carcinogens linked with lung cancer could be summed, but not risks from carcinogens affecting the lungs and, say, the liver. Another more theoretically correct approach to cancer risk assessment is to consider the weight of available evidence on hazard in defining potential risks: risks from proven human carcinogens should, in such an approach, be give more weight than risks from substances for which evidence of carcinogenicity comes only from laboratory experiments. EPA uses a five-category classification scheme (A through E) to distinguish among pollutants showing varying weights of evidence of carcinogenicity. Group A includes definite human carcinogens, Group B includes probable human carcinogens, etc., on down to Group E, which includes compounds for which there is no evidence of carcinogenicity. (EPA, 1987)

Although use of CAG dose-response values has been the standard approach throughout these studies, at least two alternative approaches have been used, or suggested, as methods of investigating the potential health effects of the urban soup: (1) the use of structure-activity relationships and (2) comparative risk methods.

Structure-Activity Methods—The Southeast Chicago study considered but rejected the use of structure-activity data as a potential means of expanding the pollutants covered in its risk assessment (Summerhays, 1987). In this approach, pollutants for which CAG has not yet developed dose-response scores are compared to CAG-scored chemicals on the basis of similar chemical structures and activities. Where chemical structures and properties are sufficiently similar, risks from the matched chemical are presumed to be the same as for the scored chemical. The benefit of this approach is to expand the number of pollutants evaluated; its obvious disadvantage is its uncertainty. Pollutants with closely similar chemical structures and activities may, in fact, have entirely different potencies.

Comparative Risk Method—The most fundamentally different risk assessment approach suggested in the studies reviewed here is to estimate the risk of a suspect carcinogen (e.g., diesel emissions), for which there are no epidemiological cancer data, by comparing the potency of the agent in short-term mutagenicity and carcinogenicity bioassays to those of known human carcinogens. The known human carcinogens are coke oven, roofing tar, and cigarette smoke tar emissions. The following formula is used:

$$\begin{matrix}\text{Estimated}\\ \text{Human Risk}_{\substack{\text{Untested}\\ \text{Mixture}}}\end{matrix} = \begin{matrix}\text{Known}\\ \text{Human Risk}_{\substack{\text{Tested}\\ \text{Carcinogen}}}\end{matrix} \times \left[\frac{\text{Bioassay Potency}_{\text{Untested Mixture}}}{\text{Bioassay Potency}_{\text{Tested Carcinogen}}}\right]$$

The IACP has used this biological approach to evaluate organic particulate emissions from specific source categories—road vehicles, residential heating (oil, wood), and utility power plants. These tests consider the relative potency in both skin tumor initiation and short-term bioassays. (Lewtas, 1987)

The advantage of this approach is that it directly evaluates complex organic mixtures (e.g., POM), potentially taking synergistic or antagonistic effects into account. This avoids potential errors in previous approaches (e.g., in the Six Months Study) of using a single compound, often B(a)P, as a surrogate for all compounds within a class (typically POM). And, because this approach uses short-term tests, potency factors can be developed more quickly and cheaply than developing human cancer data or long-term animal data. The disadvantage is that the short term tests commonly used (including in vivo skin tumorigenicity bioassays and various in vitro bioassays such as the Ames Salmonella test), evaluate mutagenicity rather than carcinogenicity. Although most known human carcinogens are also mutagens, the reverse is not true—evidence suggests that many mutagens are not carcinogens.

Comparative potency factors were used in the Motor Vehicle, 5 City Controllability, and Baltimore IEMP studies.

Noncancer effects—Noncancer effects have been considered in the IEMP Baltimore, Santa Clara, and Denver projects, but work in evaluating health endpoints other than cancer is still fairly recent and not well developed. Although some pollutants—like carcinogens—pose risks at any dose, most noncancer effects are presumed to occur only above a definable "no effect" level. Defining this level, and predicting whether this level is ever exceeded in the ambient environment, is the focus of noncancer health effects evaluations of the urban soup.

Although few studies have attempted to assess noncancer endpoints, those which do must consider the following issues:

- Noncancer effects are assumed *not* to be additive for dissimilar endpoints: Where noncancer effects have been demonstrated, their range of nature and severity is wide. Some effects are temporary and reversible (headaches, blurred vision); others permanent but nonfatal (birth defects, neurological disorders); and some fatal or potentially fatal (heart disease). It is clearly inappropriate to sum across such different effects.

- Threshold values for noncancer effects may be time-sensitive: For some pollutants, adverse health effects are only assumed to exist if the threshold is exceeded for long periods of time (e.g., one year). For others, the significant exceedance period may be minutes or hours. Very few data are available on which to ground reasonable assessments of effects.

- Thresholds may vary among individuals or across population groups: For instance, lead has much more severe effects on children than on adults. Individual sensitivities can change over time—sensitization to formaldehyde is an example.

- Dose-response relationships above threshold doses are a separate factor for evaluation: When thresholds are exceeded, dose-response relationships can follow different curves—linear, step-function, curves of various shapes.

Studies that have addressed these issues typically make a number of simplifying assumptions. The most common is to focus on the threshold level alone, disregarding the shape of the dose-response curve above the threshold. The statistic of interest is simply whether or not a substance exceeds its threshold or some fixed fraction of its threshold, such as 10 percent or 25 percent—the latter approach leaving room for other routes of exposure, such as residential indoor air, occupational exposures, or noninhalation routes. As an example, the Santa Clara IEMP estimated that there were 100,000 people in the study area at some risk of blood effects from benzene exposures.

Health data on noncarcinogens are limited. Although industrial exposure standards have been established for many chemicals, EPA has not endorsed their use for ambient air toxics studies for a number of reasons, one of which is because they are usually based on the acute effects of relatively high exposures to healthy adult male populations in work settings. Effects of involuntary, long-term exposures at subacute levels have not been evaluated in equivalent detail, particularly on potentially more vulnerable subpopulations. EPA is in the process of developing inhalation reference doses, which will facilitate the inclusion of noncarcinogenic effects in future studies.

The IEMP Santa Clara, Baltimore, and Denver projects have, however, provided limited data on noncarcinogenic evaluations of the urban soup.

Santa Clara—Because the only available noncarcinogenic potency scores available at the time were based on using oral reference doses as a surrogate for inhalation values, the Santa Clara study stopped short of actually doing a risk assessment for noncarcinogens. Instead, the percentages of the population above the threshold were estimated for the following effects: immune system, blood, liver, and kidney. These estimates were based solely on chronic effects because only annual average modeled concentrations were available, and because of data limitations for short-term health effects.

It is possible that the exceedances of thresholds that were identified in the Santa Clara study could be translated into cases of noncarcinogenic risks in the future once more complete coverage of inhalation reference doses is available.

Baltimore—Noncancer effects were determined in two ways. First, the increased risks of several noncancer effects (liver toxicity, kidney toxicity, reproductive, neurological, fetal, and blood) were evaluated for specific compounds in the modeling exercise. In particular, the effect threshold(s) relevant to each pollutant was divided into the model-predicted ambient air concentration of that pollutant. If the resulting ratio exceeded unity, a concern for noncancer effects was identified. Second, a hazard index (discussed earlier in this chapter) was developed that summed individual pollutant ratios by effect category. This latter analysis was aimed at examining the impact of exposure to complex chemical mixtures in the ambient air: if the hazard index exceeded unity, the concern for noncancer effects would be the same as for the exceedance of a threshold value. The resulting analysis suggested, preliminarily, some concern for blood effects from benzene exposures and elevated concern for the following effects from xylene exposures: liver, kidney, reproductive, neurological, fetal, and blood.

Denver: The IEMP Denver study collected 12- and 24-hour measured air quality data on days that include limited coverage of peak pollution levels. The study anticipated that the data gathered on peak exposure days may be of future use when additional short-term health effects data become available. The project will evaluate these effects to the extent feasible with currently available data, but is deliberately gathering more data than it can now evaluate in expectation of better future information—an interesting precedent for other studies.

5.3 Comparison of Approaches to Interpreting Exposure and Risk

Table 5-1 compares the emphasis of the various studies in relation to interpreting exposure or risk information.[4]

Table 5-1. Comparison of Approaches to Interpreting Exposure and Risk Information

Study	Population Exp or Risk	Maximum Exp or Risk	Rank Exp or Risk, by Source	Rank Exp or Risk, by Pollutant	Evaluate Patterns of Exp or Risk
South Coast MATES	X	X	Stat. v. Mobile	X	X
Motor Vehicle Study	X			X	
Clark County	X			X	
Southeast Chicago	X	X	X	X	X
IEMPs					
Philadelphia	X	X	X	X	X
Baltimore	X	X	X	X	X
Santa Clara	X	X	X	X	
Kanawha	X	X	X	X	X
Denver	X		Area sources	X	
NESHAPS*	X			X	
35 County Study*	X	X	X	X	
5 City Controll. Study	X		X	X	X
Urban Air Toxics Program	Goals vary	Goals vary	Goals vary	Goals vary	Goals vary
TEAM	X	X		X	X
IACP	X				

* Part of Six Months Study

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[4] This information is taken, in some cases, only from published final reports and may therefore not capture all analyses conducted for each study.

The purpose of this report is to present methods and approaches, not results. The brief discussions below therefore cover only the most general types of exposure and risk conclusions, emphasizing problems of interpretation rather than statements of results.

Ranking the Relative Importance of Pollutants

Most studies evaluated the relative importance of pollutants in terms of potential population risks. An important general issue in ranking the relative significance of pollutants has been the accuracy of potencies or the assignment of default potencies to certain classes of pollutants. For instance, in the absence of data on the speciation of airborne chromium between the hexavalent and the trivalent forms, some studies have assumed, conservatively, that all chromium emissions are in the hexavalent form. This significantly increases the calculated risks associated with chromium. Actual chromium risks are likely far lower, since a substantial portion of emissions likely occur in the trivalent form. The Santa Clara IEMP assumed that only 10 percent of ambient chromium is Cr+6. The 5 City Controllability Study avoided this problem by directly estimating both Cr+6 and total chromium emissions and modeling them separately. Preliminary results from the 5 City Study suggest that only about one-third of total chromium emissions will be as hexavalent chromium.

The problem of speciation has occurred with beryllium, nickel, and products of incomplete combustion (PICs), where some investigators have assigned all species of a compound with the toxicity of a particular compound or chemical form for which data are available.

Ranking the Relative Importance of Sources or Source Categories

A general conclusion of a number of the studies reviewed was that area source-related pollutants tend to dominate population-weighted average risks, but that point sources often dominate MEI impacts. Within area sources, mobile source-related compounds—generally POM and VOC—tend to dominate exposures and risks. Wood stoves, fireplaces, chrome platers, comfort cooling towers, hospital sterilizers, small degreasing operations, and gasoline marketing are other area source contributors to exposures and risks in most urban areas.

The Santa Clara study investigated risks from some relatively unusual classes of sources, such as volatilization of organics from drinking water treatment plants and electronic component manufacturing. Risks from these sources were found to be relatively minor in comparison with more common sources of toxic air exposure. This study also did some risk analysis of highways to evaluate MEI exposures near intersections.

For some point source emission categories, such as those investigated in detail in the Kanawha Valley study, the general conclusion has been that fugitive and vent releases may often dominate air toxic risks, in large part because of their release characteristics. Since emissions from these sources are at low elevations, potential exposures and risks near the sites may be relatively high. Stack emissions of a particular air toxic tend to pose lower risks than low level emissions of the same compound because greater dispersion is likely to occur prior to exposure.

One "source" that has not been addressed in significant analytical detail is background contributions from adjoining regions. As noted in Chapter 4, modeled exposures tend to be consistently lower than monitoring concentrations of the same pollutants. The inability of modeling studies to consider background is one of several factors contributing to this underestimation. Transport may be most important for long-lived substances such as carbon tetrachloride, but it can theoretically be a significant factor for other pollutants as well.

Evaluating Exposure and Risk Patterns

Comparisons of MEI risks to average individual risks were done in the IEMP studies (especially Philadelphia and Kanawha). On the basis of the types of evidence developed by these programs, it appears that the ratios of MEI to average risks (or concentrations) may be significant in some areas, possibly between 10 and 100. This ratio, however, is strongly dependent on how close residential areas are to major industrial facilities or other significant point sources. In Philadelphia, the nearest residences to industrial facilities were typically much farther away than they were in the Kanawha Valley. In relatively unindustrialized areas, such as Denver, ratios of MEIs to average population exposures will typically be considerably lower.

Using TEAM data, it is possible to look at ranges of average population exposures. TEAM studies suggest that maximum exposures within a population are likely to be far higher than mean exposures, possibly by as much as three or more orders of magnitude. Figure 5-2 presents examples of the wide range in concentrations observed based on personal sampling. (Wallace, 1987) These plots show the differences in indoor and outdoor concentrations, based on matched-pair TEAM samples, for several important VOCs. (The matched-pair values actually represent the differences between personal monitoring samples and outdoor samples; however, since the samples represent the 7p.m.-7a.m. time frame, the personal monitoring samples largely reflect indoor concentrations.) All outdoor sampling was conducted in direct proximity to the indoor sampling locations.

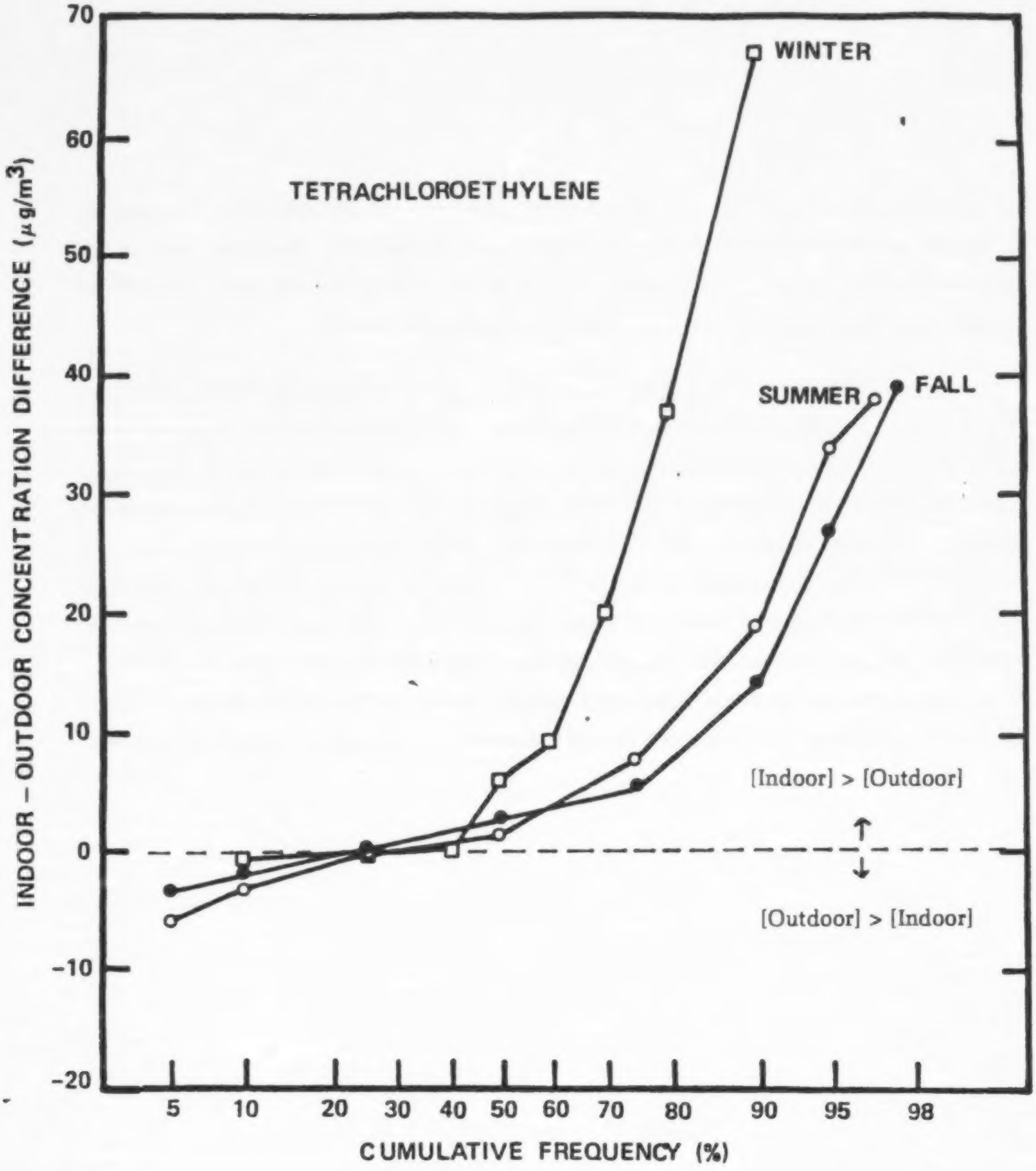

Figure 5-2. Distributions of Matched Pair Concentrations Differences, Based on Indoor and Outdoor TEAM Samples, from 7pm to 7am

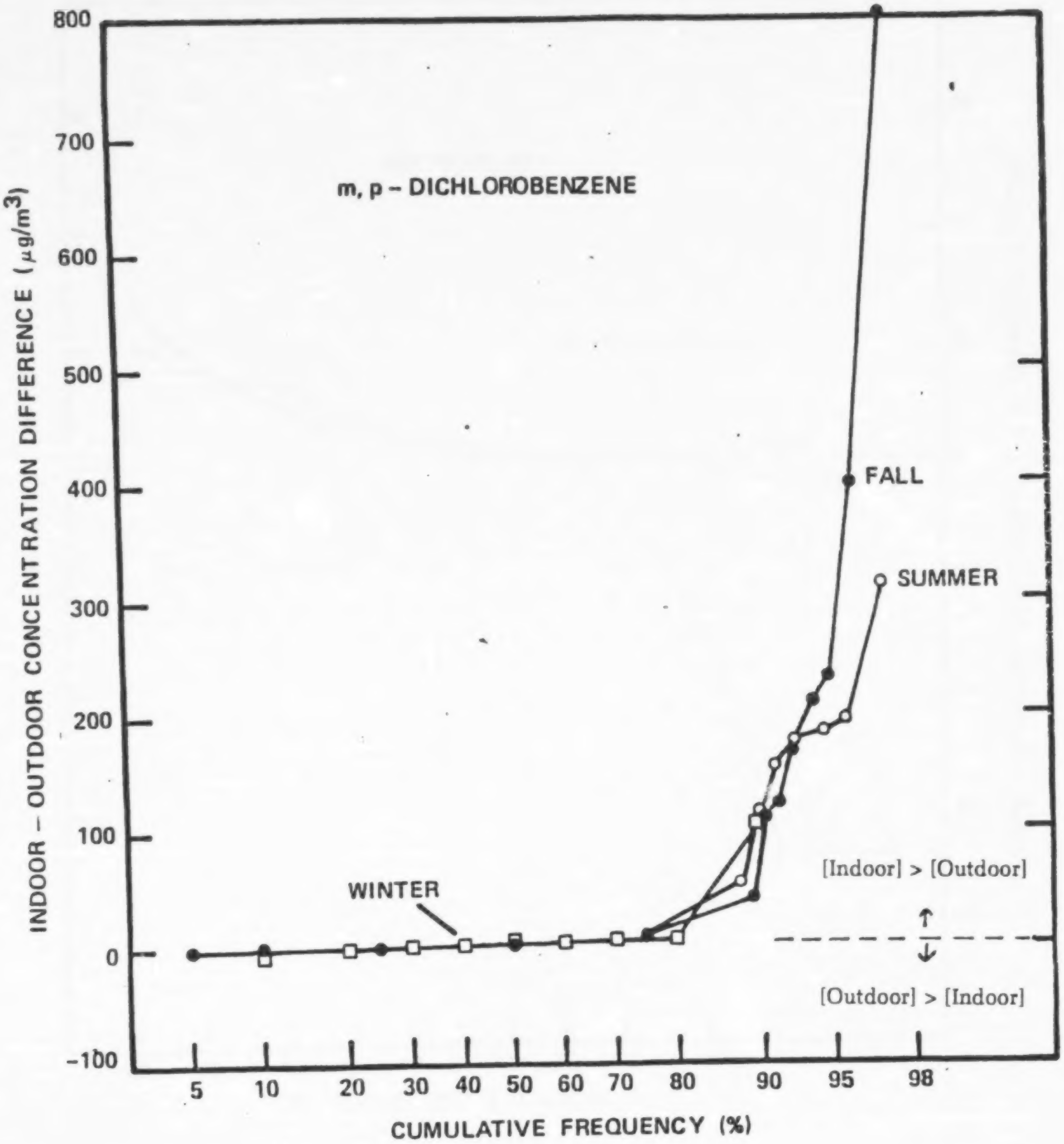

Figure 5-2 (cont'd)

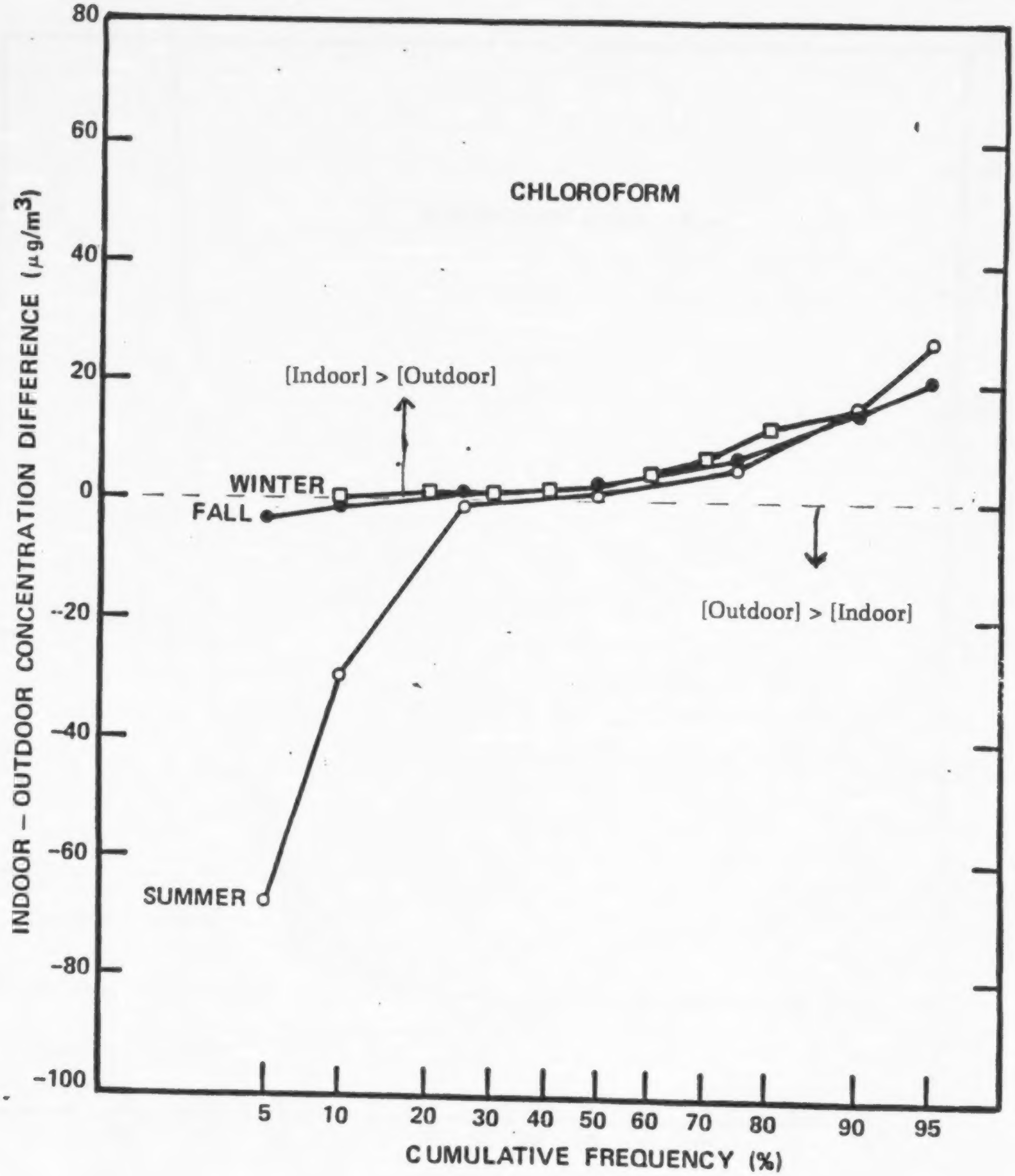

Figure 5-2 (cont'd)

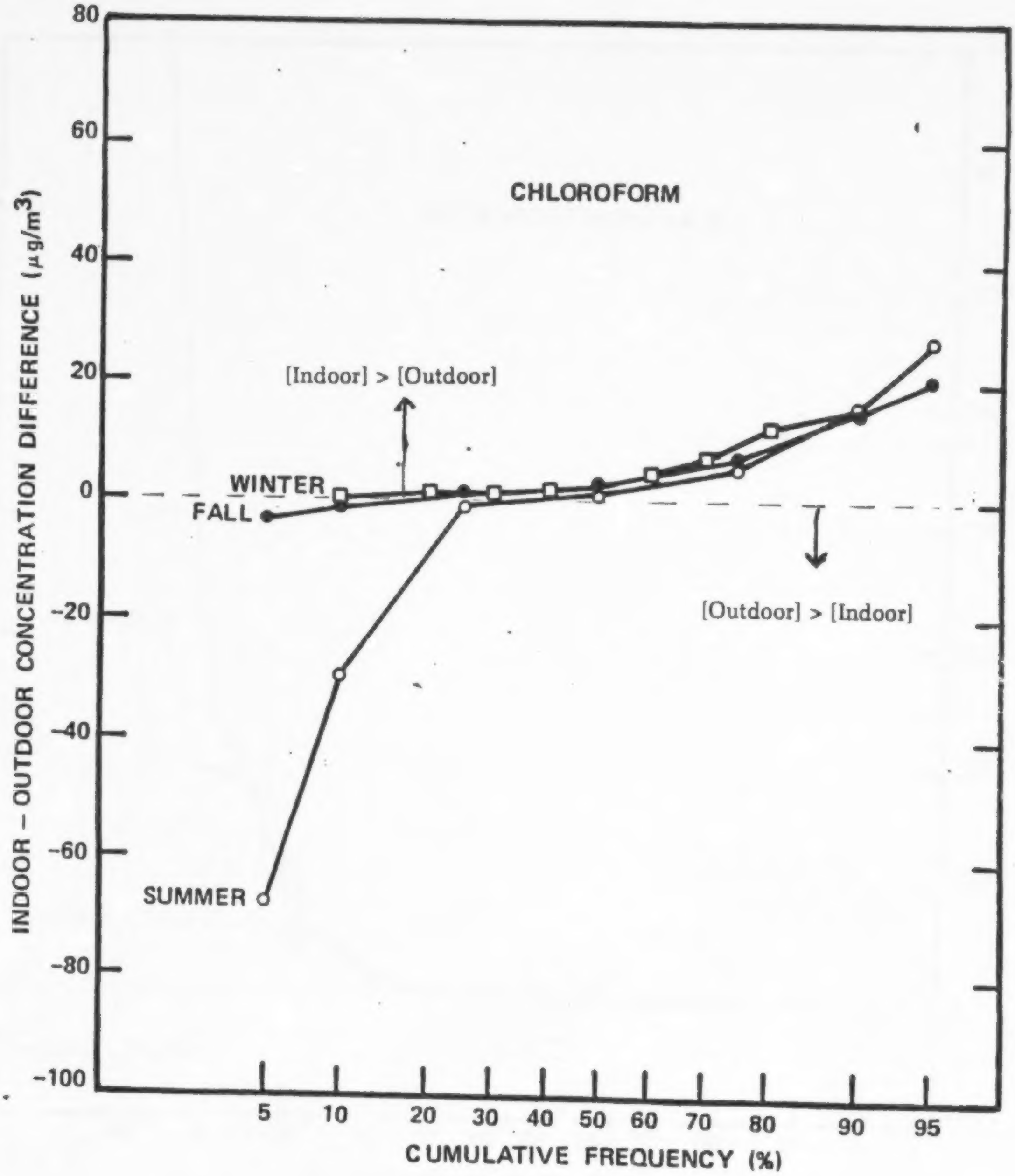

Figure 5-2 (cont'd)

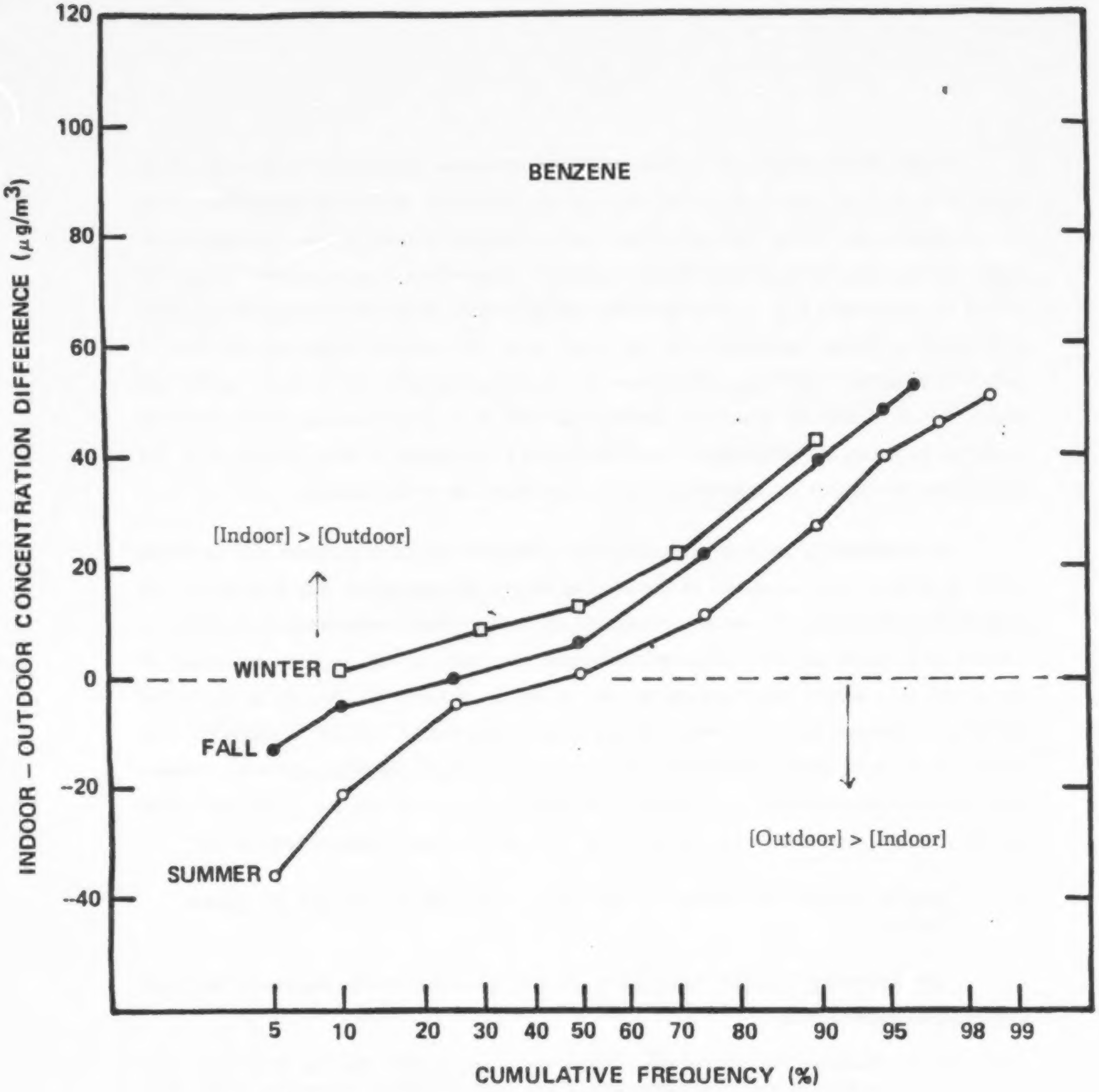

Figure 5-2 (cont'd)

Since TEAM compares outdoor ambient exposures with indoor exposures of all types (occupational exposures, consumer product exposures, residential exposures), it is not surprising that some fraction of the indoor sample concentrations exhibits much higher levels than the matched outdoor samples. This effect is pronounced during the winter for pollutants (e.g., p-dichlorobenzene) associated with consumer products (e.g., mothballs) or indoor activities. On the other hand, the middle range and low end of indoor and outdoor exposures documented by TEAM tend to be in much closer agreement. Again, this is intuitively plausible—ambient air will be a proportionally more important source of exposure for populations experiencing low workplace or other indoor air toxics concentrations, in part because of infiltration of outdoor air to the indoors.

In investigating patterns of exposure near large industrial complexes, the Kanawha study apparently was successful in the level of source disaggregation that it selected. As discussed in Chapter 4, this study represented emissions within large chemical facilities as a series of grouped sources. Alternatives would have been, at one extreme, to model all emissions as a single point source at the centroids of each facility, or, at the other extreme, to attempt to model every release point individually. Analysis suggested that results would have been significantly less accurate, especially for MEI exposures, if these large complex sources had been modeled as single points, but that not much additional accuracy, if any, would have been gained if all sources had been modeled individually.

5.4 Insights into the Use of Exposure and Risk Assessment in Multiple Air Toxics Studies

The following insights have been gained from air toxics exposure and risk assessments done to date.

- Perhaps the single most important insight that can be offered is that exposure and risk assessments are very complex, integrating many data bases, procedures, and assumptions. As such, the user should be very careful in interpreting the results or comparing the results to those of another study. Major uncertainties are involved in all steps of the process, from monitoring, emission inventorying, modeling, assigning exposure levels to populations, and attributing health effects to exposure levels.

- Because of the errors, uncertainties, assumptions, and limitations involved in risk assessments, most investigators concede that their results should not be interpreted to represent absolute or precise predictions of cancer or other health risk. At best, most studies have labeled themselves as screening or scoping studies, only yielding estimates of the *relative importance* of various sources, pollutants, etc., or the *relative merits* of alternative regulatory measures to control air toxics.

- Because cancer unit risk factors are considered to be conservative for a number of reasons, some studies have concluded that their results are biased conservatively high. In fact, there are other potential biases in these studies, some of which may lead to underpredictions of cancer risk. Missing source categories, unaddressed pollutants, underpredicting models, and underaccounting of pollutant transformation, all represent potential biases on the low side. Conservative unit risk factors, conservative exposure assumptions, and the assumption of additivity among different carcinogens all represent potential biases on the high side. The practice of considering all chromium, beryllium, and nickel to be as carcinogenic as particular forms of these metals (i.e., Cr+6, beryllium sulfate, and nickel carbonyl/subsulfide, respectively) also causes significant biases on the high side, where this is done.

- The use of EPA's Human Exposure Model frees the study manager from having to associate concentration data with population data in an exposure assessment, as this is done internally using stored 1980 U.S. Census Bureau data. This is a potential advantage for areas not wanting to develop their own exposure assessment modeling capabilities.

- Evolving methods being developed by the TEAM and IACP studies both challenge the "standard" assessment methods used in most studies to date, and offer promise for improved methods in future studies. For example, the TEAM results challenge the standard assumption of constant exposure to ambient outdoor air, and suggest that personal exposures may be much more influenced by indoor, workplace, and product exposures than by ambient air in many cases. As another example, the IACP challenges the practice of evaluating pollutants individually, and is developing techniques to access the effects of complex mixtures through the use of source apportionment/bioassay fractionization of ambient samples and the development of comparative potency factors for complex organic mixtures.

- The study manager should be aware that the data, techniques, and assumptions in the field of cancer risk assessment have changed rapidly in the past five years and will probably continue to be dynamic as better data become available. Changes in key emission factors and cancer unit risk factors, for example, can dramatically change both the absolute and relative contributions of particular pollutants, source categories, etc., in one's study analysis. The study manager should thus either maintain a dynamic data base, incorporating changes as they are perceived, or be prepared to be second-guessed if he/she "freezes" the study at a certain point in time and certain data elements, assumptions, or techniques become outdated.

Chapter 5

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CHAPTER 6

CONTROL STRATEGY SIMULATION AND EVALUATION

The following topics are discussed in this chapter:

- The use of control strategy simulation and evaluation in urban air toxics assessments
- Comprehensive vs. site-specific strategy simulation
- Control strategy simulation procedures in the 5 City Controllability Study
- Insights on control strategy simulation and evaluation

6.1 The Use of Control Strategy Simulation and Evaluation in Urban Air Toxics Assessments

A primary objective of urban air toxics assessments has been to define the existing levels of emissions, concentrations, exposures, and risks in the study area. This has generally involved the assessment of current conditions in what is often termed "baseline" or "base year" analyses. Some studies have stopped at this point. Ambient air monitoring studies can only assess current conditions because of their inherent inability to project future ambient air levels. Emission inventory/dispersion modeling studies, on the other hand, are suitable for control analyses, because emissions can reasonably be projected into the future both as a function of anticipated growth in an area and as a function of alternative control measures that may be applied. As indicated previously, this represents a significant advantage of using an emission inventory as a basis for conducting an urban air toxics assessment.

A principal objective in several of the studies reviewed was the analysis of the potential for risk reductions achievable through alternative control measures. To do this, specific control measures or combinations of control measures are superimposed on the base case emission inventory, and the resulting emission projections are then modeled in the same manner as in the base year analysis to estimate reductions in exposure and risk. Thus, the emission (inventory) projection becomes an important tool in carrying out the control strategy evaluation.

Once the control analysis is completed, the study manager can provide the local policymakers (i.e., risk managers) with information needed to prioritize various alternative measures, based on risk reduction potential. Cost-effectiveness can also be evaluated. A cost-effectiveness analysis of control options can generate information highly useful to local policymakers in setting control priorities. Decision makers can get a sense of the effectiveness of individual controls and total control strategies by seeing their costs to reduce cancer risk, either in terms of aggregate (population) incidence or risk to the maximum exposed individual. This allows them to allocate their community's limited resources into activities that provide the greatest environmental health protection.

Such analyses can also yield estimates of "co-control" potential, i.e., the extent to which measures designed for air toxics control will also control ozone, PM-10, or other criteria pollutants. This is an important consideration as air toxics control benefits may help "sell" certain criteria pollutant control measures that may otherwise have marginal acceptability on their own merits. Similarly, the concept of co-control may help the acceptance of certain air toxics measures.

6.2 Comprehensive vs. Site-specific Strategy Analyses

Two types of control strategy analyses are employed in the studies reviewed. One can be best described as a comprehensive or "across-the-board" type of analysis, wherein different combinations of controls are applied (hypothetically) to all facilities within many source categories. This type of analysis was employed in the 5 City Controllability Study. The second type of analysis is more limited and site specific. In this latter type of analysis, the technical feasibility of candidate control measures is first evaluated in detail for particular facilities within the study area, followed by a simulated application of these specific measures to determine specific exposure and risk reductions. The IEMPs employed the more focused, site-specific type of analysis (to varying degrees).

In both types of analyses, emissions projections are made corresponding to the control scenarios simulated. Fundamental differences are due to (1) the extent of source coverage, (2) the consideration of multiple vs. single control strategies, and (3) the incorporation of growth and plant retirement into the future projections of emissions and risk.

Extent of Source Coverage

The comprehensive coverage in the 5 City Study allowed for the simulated application of controls to all sources within any source category to which a given regulatory option may apply. For example, the 5 City Controllability Study could simulate the incorporation of drift eliminator retrofits on all industrial cooling towers or scrubbers on all hospital sterilizers within each study area, without having to identify specific candidate facilities for analysis. In contrast, the control measures superimposed in the IEMPs typically targeted a handful of source categories or specific facilities for analysis, albeit categories thought to be more important.

Multiple vs. Limited Control Options

In the 5 City Controllability Study and Philadelphia IEMP, multiple control options were analyzed for the sources under consideration. The idea behind this analysis was to provide an assessment of different combinations of potential measures, from both a technical and a cost standpoint. In contrast, the Santa Clara IEMP considered only one (or two) control options per source type based on a preliminary decision of what appeared to be the most cost-effective.

Inclusion of Source Growth and Retirement

Of the studies reviewed, only the 5 City Controllability Study projected emissions and risks to a specific year in the future. This study used 1980 as its base year and 1995 as the year to which projections would be made. This time interval was considered long enough that new source growth and old source retirement had to be factored into the analysis. This added considerable complexity to the analysis, as growth and retirement rates had to be determined for each source category undergoing analysis, and separate control efficiencies had to be considered for existing sources vs. new growth and replacement growth sources.

6.3 Control Strategy Simulation Procedures in the 5 City Controllability Study

The 5 City Controllability Study is both an analysis of existing conditions in a "base year" (defined as 1980) and an analysis of conditions in a projection year (1995), reflecting alternative control scenarios. The heart of this study is the base year inventory and a regulatory impact model (RIM) that operates on the base year inventory to simulate different combinations of control measures. Emission projections made by RIM are subsequently used in EPA's Human Exposure Model (HEM) to estimate reductions in cancer incidence.

RIM Operation

Figure 6-1 shows a schematic of the Regulatory Impact Model (RIM) that is used to estimate future emissions and costs of emission control. As shown in Figure 6-1, there are two functional components of RIM: an emissions projection module and a control cost module.

Operation of RIM starts with the baseline emissions inventory which, for each source type, contains information relating to annual emissions and the existing level of control in place on each source type. From this inventory, the uncontrolled emission rates can be determined by back-calculation for the base year.

To project changes in the baseline emissions to any future year, three pieces of information, specific to each source type, must be established:

1. The rate at which old equipment will be replaced with new, less polluting equipment;

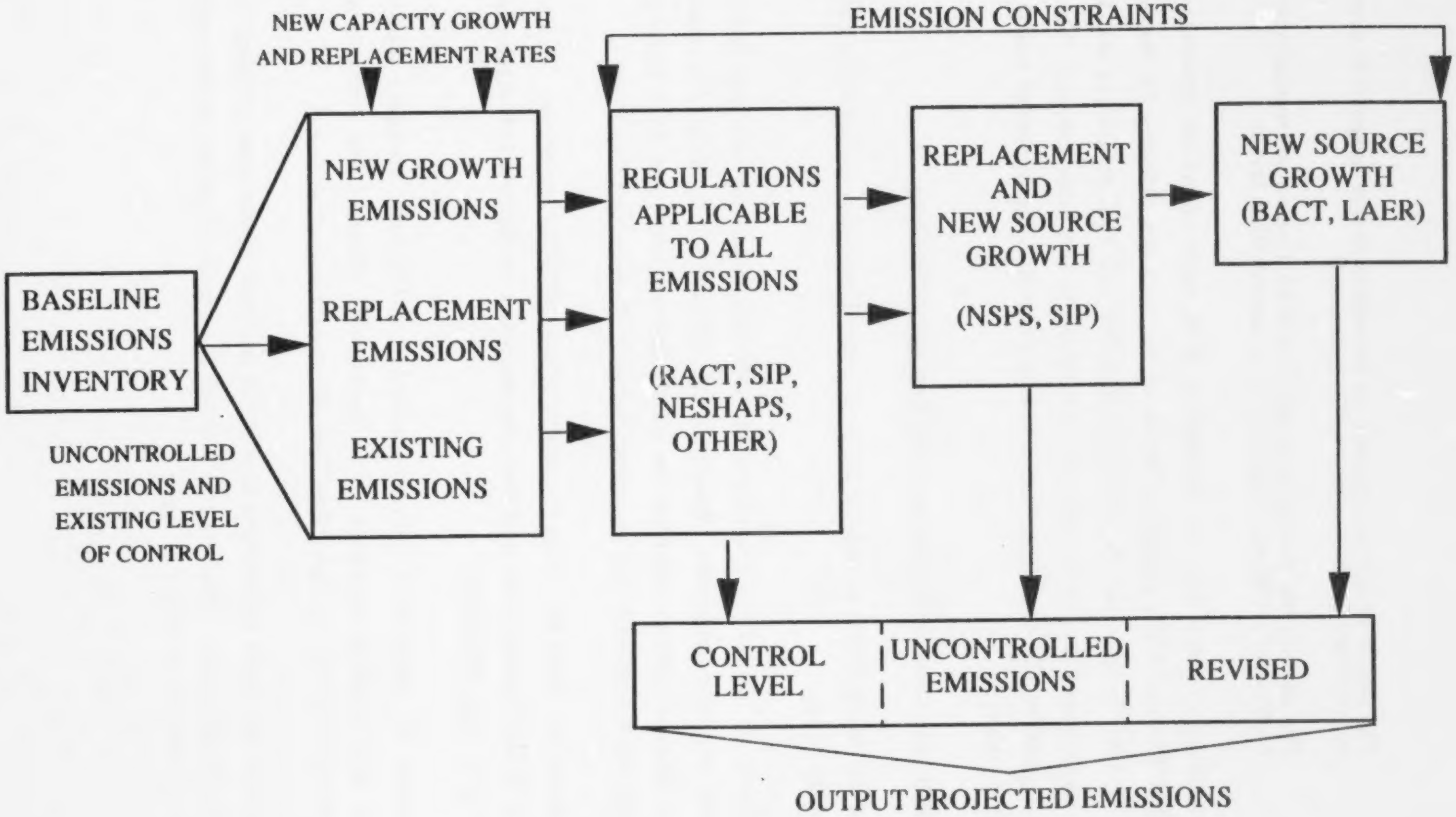
NEW CAPACITY GROWTH AND REPLACEMENT RATES
EMISSION CONSTRAINTS
BASELINE EMISSIONS INVENTORY
UNCONTROLLED EMISSIONS AND EXISTING LEVEL OF CONTROL
NEW GROWTH EMISSIONS
REPLACEMENT EMISSIONS
EXISTING EMISSIONS
REGULATIONS APPLICABLE TO ALL EMISSIONS
(RACT, SIP, NESHAPS, OTHER)
REPLACEMENT AND NEW SOURCE GROWTH
(NSPS, SIP)
NEW SOURCE GROWTH (BACT, LAER)
CONTROL LEVEL
UNCONTROLLED EMISSIONS
REVISED
OUTPUT PROJECTED EMISSIONS

Figure 6-1 Schematic of Regulatory Impact Model (RIM) Used in 5 City Controllability Study

2. The rate at which the industry (or emissions source category) is expected to experience growth in a geographic region; and

3. The constraints that existing and future environmental regulations impose on sources to reduce emissions from uncontrolled levels.

Accordingly, three files are created in RIM, each of which operates on the uncontrolled PM and VOC emission levels of each source category in the baseline inventory. (Note: In this analysis, RIM calculates PM and VOC reductions and assumes that particulate toxics and toxic VOC are controlled to the same extent.) The actual calculations made by RIM cannot be described here in detail. The interested reader should consult (EPA, 1985).

6.4 Control Strategies Evaluated in 5 City Controllability Study

The following listing shows the kinds of control strategies evaluated in the 5 City Controllability Study.

Scenario 1: Emissions projections for 1995 under existing and expected criteria and NESHAP regulatory programs. Results of this scenario are considered representative of the 1995 emissions picture assuming the anticipated regulatory agenda is accomplished. This scenario would reflect toxics co-control benefits of ozone and PM-10 SIPs.

Scenario 1a: Same as 1, plus the effects of new NESHAP initiatives. This scenario might result if EPA focuses control of toxic air emissions on Section 112 of the Clean Air Act, resulting in more NESHAPS.

Scenario 2: Same as 1, with the addition of the most stringent (reasonable) controls on new capacity emissions. The incremental effect of this scenario may be described as requiring very stringent BACT on all new sources of air toxics.

Scenario 2a: Same as Scenario 2, with the addition of the most stringent controls on road vehicle emissions. This scenario imposes the control of mobile source emissions required in Los Angeles to all study areas.

Scenario 2b: Same as 2a, with the addition of the most stringent controls (reasonable) on replacement emissions. The scenario extends stringent BACT to replacement sources of toxic air emissions.

Scenario 3: Same as 2b, with most stringent (reasonable) control on all (i.e., new, replacement, existing, and retrofit) emissions. This may best represent a requirement for stringent BACT on all air toxics sources.

6.5 Assumptions Inherent in 5 City Controllability Study Control Analysis

A number of important assumptions had to be made in the 5 City Controllability Study in order to complete the analysis. Assumptions that are made in a broad national scoping study, because of insufficient resources to analyze individual facilities/pollutants/controls, may not be acceptable to some policymakers who need a firmer basis for taking regulatory action. Many of these assumptions, listed below, should be reviewed carefully in more detailed, locale-specific studies.

- The baseline emissions and control efficiency data are assumed to be accurate in the existing inventory. Many of the projection algorithms are based on these two parameters, so it is essential that they be as accurate as possible. Because uncontrolled emissions are back-calculated from existing controlled emissions using the control device efficiency levels, this type of analysis is very sensitive to errors in the control device efficiency, especially as it approaches 100 percent.

- Control levels of toxic organics and toxic PM are assumed proportional to control levels of VOC and PM, respectively. This assumption should be challenged in a more detailed study, at least for the more important sources and pollutants. Some metals, for example, may be present in a gaseous state in a high temperature exhaust and may not be controlled at the same level as PM from some control hardware.

- Both new and replacement growth, as well as capacity retirement, are assumed to occur at the same location as existing sources. A State or local agency may have more detailed, plant-specific data that could obviate the need to make projections in this manner. The locations of large plants expected to open or close in a few years is probably well known to local officials, so growth and retirement could probably be handled without having to make general projections based on published trends data.

- Control measures are assumed to be applicable to the targeted sources, without regard for technical feasibility on a case-by-case basis. Although the measures selected in this study are generically applicable, there may be specific technical reasons why they would not be appropriate in certain facilities. Again, a State or local agency would be in a better position to make case-by-case projections.

6.6 Control Strategy Evaluation in the IEMPs

The approach for simulating and evaluating controls in the IEMPs was much more focused on specific sources, pollutants, and control measures than in the 5 City Controllability Study discussed above. The Philadelphia and Baltimore IEMPs were somewhat more site specific than the Santa Clara IEMP. All are discussed separately below.

Philadelphia IEMP Control Analysis

In the Philadelphia study, controls were evaluated for a "cluster" of point sources centered in a particular section of the city. Questionnaires and phone calls to industry were used to help tailor the control options to specific space, power, and cost constraints at each plant site. Even so, the analysis provided only a rough estimate of costs and effectiveness because a more comprehensive engineering analysis was beyond the scope of the study. A total of eight point sources were evaluated. Control options included charcoal adsorption, solvent substitution, steam stripping, and scrubbers. Costs and emission rates to all media were estimated to address risk reductions on a multimedia basis.

The Philadelphia IEMP study also evaluated controls for several area sources including dry cleaning, gasoline marketing, degreasing, and miscellaneous solvent usage. Control options included vapor recovery, solvent substitution, and Stage I and Stage II controls for gasoline marketing. The feasibility of each option was evaluated, as well as the cost and pollutant removal effectiveness.

Table 6-1 summarizes the control options evaluated in the Philadelphia IEMP. Multiple options were analyzed for each facility or source category and then different combinations of options were evaluated for cost-effectiveness. Table 6-2 shows the projected cost-effectiveness of various control simulations in terms of reducing annual, aggregate cancer incidence.

Table 6-1 Control Options Evaluated in the Philadelphia IEMP

Source	Control Option	Control Descriptions
Degreasing	Option 1	Cold Cleaners: cover during idle time (90 percent control efficiency); drain racks with 30-second drains (50 percent control efficiency) Open Top Vapor Degreasers: cover during idle time (90 percent control efficiency); increase freeboard ratio during operation (27 percent control efficiency)
	Option 2	Cold Cleaners: cover during idle time (90 percent control efficiency); drain racks with 30-second drain (50 percent control efficiency) Open Top Vapor Degreasers: cover during idle time (90 percent control efficiency); use refrigerated freeboard device (60 percent reduction in vapor losses, 29 percent reduction of carry-out losses)
	Option 3	Cold Cleaners: cover during idle time (90 percent control efficiency); drain racks with 30-second drain (50 percent control efficiency) Open Top Vapor Degreasers: cover during idle time (90 percent control efficiency); carbon adsorber (reduce 70 percent vapor losses, 30 percent carry-out losses)
Refinery B	Option 2	Convert noncontact floating roofs to contact floating roofs on benzene tanks only; leak detection and repair methods (air)
	Option 3	Secondary seals on benzene tanks only; install rupture disks for controlled degassing reservoirs for compressors (air)
	Option 4	Convert noncontact floating roofs to contact floating roofs for benzene tanks only; install rupture disks for safety/relief valves and seals with controlled degassing reservoirs for compressors (air)
	Option 21	Secondary seals on benzene tanks only; install rupture disks for controlled degassing reservoirs for compressors; convert noncontact floating roofs to contact floating roofs for benzene tanks only; install dual mechanical seals with a barrier fluid system and degassing reservoir vents on light liquid pumps (air)
	Option 23	Secondary seals on benzene tanks only; install rupture disks for controlled degassing reservoirs for compressors; install a thermal oxidation system on all benzene and gasoline tanks; install dual mechanical seals with a barrier fluid system and degassing reservoir vents on light liquid pumps; require more frequent inspection on the valves and addition of sealed bellow valves (air)

Table 6-1 Control Options Evaluated in the Philadelphia IEMP (cont'd)

Source	Control Option	Control Descriptions
Other Industrial	Option 1	Enlarged condensation zone, waste recovery facility, manual enclosure (air)
	Option 2	Same as Option 1, except automatic enclosure
Industrial Dry Cleaner	Option 1	Inspection and maintenance
Queen Lane and Belmont Drinking Water Treatment Plants	Option 1	Granular activated carbon (GAC) high-effectiveness
	Option 2	Granular activated carbon (GAC) medium-effectiveness
	Option 3	Granular activated carbon (GAC) low-effectiveness
Baxter Drinking Water Treatment Plant	Option 2	GAC high-effectiveness
	Option 3	GAC medium-effectiveness
Dry Cleaning	Option 1	Inspection and maintenance (air)
	Option 2	Same as Option 1, plus carbon adsorption for commercial dry cleaners (air)
	Option 3	Same as Option 2, plus carbon adsorption for coin-op and commercial dry cleaners (air)
Chemical Manufacturer	Option 1	Carbon adsorbers on each of three vents (PCE emissions)
	Option 2	Carbon adsorbers on each of three vents (DCP emissions)
	Option 3	Option 2 plus Option 1 (air)
	Option 4	GAC of DCE and DCP waste streams (water)
	Option 5	Steam stripping of DCE and DCP waste streams (water)
	Option 10	Option 2 plus Option 4
	Option 11	Option 2 plus Option 5
Refinery A	Option 1	Secondary seals on internal floating roofs of gasoline tanks (air)
	Option 2	Install secondary seals on internal floating roof gasoline tanks only; leak detection and repair methods (air)
Gas Marketing	Option 1	Stage II, no enforcement inspection (air)
	Option 2	Stage II, plus enforcement inspections (air)
	Option 5	Option 2 plus onboard controls on all vehicles

Table 6-2. Estimated Cost Effectiveness of Philadelphia IEMP Control Options

PHASE II RESULTS INTENDED FOR POLICY DEVELOPMENT[1]

SCHEDULE OF CONTROL OPTIONS FOR REDUCING ANNUAL CANCER INCIDENCE

Air

(1986 analysis)

Cases Reduced per Year	Percent Reduction in Cancer Incidence from Current Control	Total Cost ($1,000/year)	Average Cost per Case Reduced from Current Control ($1,000/case)	Incremental Cost per Incremental Case Reduced ($1,000/case)	Pollution Controls Implemented[2]	
					Source	Control Option
0.0	0.0	0	0	0		
0.03	8.4	-122	-3,719	-3,719	Industrial dry cleaner	1
					Other industrial	1
					Refinery B	2
					Degreasing	2
0.09	24.1	323	3,435	7,417	Industrial dry cleaner	1
					Other industrial	1
					Degreasing	2
					Refinery B	3
					Dry cleaning	2
0.15	39.7	632	4,074	5,150	Industrial dry cleaner	1
					Other industrial	1
					Refinery B	2
					Degreasing	2
					Chemical manufacturer	5
0.19	49.9	900	4,621	6,700	Industrial dry cleaner	1
					Other industrial	1
					Refinery B	2
					Degreasing	3
					Dry cleaning	2
					Chemical manufacturer	5
0.25	63.4	2,080	8,402	19,667	Industrial dry cleaner	1
					Other industrial	1
					Degreasing	3
					Dry cleaning	2
					Refinery B	23
					Refinery A	2
					Chemical manufacturer	10
0.30	77.1	6,726	22,335	92,880	Industrial dry cleaner	1
					Other industrial	2
					Degreasing	3
					Refinery B	22
					Dry cleaning	2
					Gasoline marketing	5

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1 The unit risk factors used in this analysis are based on conservative assumptions that generally produce upper bound estimates. Because of limitations in data and methods in several areas of the analyses, such as exposure calculations and pollutant selection, risk estimates were calculated as aids to policy development, not as predictions of actual cancer risks in Philadelphia. Actual risks may be significantly lower; in fact, they could be zero. The proper function of the estimates is to help local officials select and evaluate issues, set priorities, and develop control strategies for the topics examined.

2. See Table 6-1 for definition of control options.

CAUTION: All emission and risk reductions and cost estimates in this table are shown only for example purposes, and are not intended to apply in other situations and locales.

Baltimore IEMP Control Analysis

The Baltimore IEMP performed a control analysis for four point sources. Sources and pollutants were selected for the control options analysis if they contributed 1 percent or more of the cancer incidence in the study area, or more than five in a million average individual risk within a particular grid cell. Each facility emitted metals (most notably hexavalent chromium) and POMs, with one plant also being a major source of benzene emissions. Emission reductions and associated costs were determined for baghouses and for air toxics co-control benefits of VOC and particulate controls. Air pollution controls were evaluated on a risk reduction basis.

The study identified potentially available point source controls by first consulting with Maryland Air Management Administration (AMA) staff responsible for monitoring permit compliance of the respective facilities. Once potential control options were identified, costs and removal efficiencies were calculated based on data in the literature, input from AMA staff, and estimates provided by vendors.

Performing the site-specific assessment needed to accurately quantify control costs and removal efficiencies for complex industrial facilities was beyond the scope of this study, since neither the time or resources were available to identify and accurately evaluate individual control options. The intent of the study was to estimate relative costs and effectiveness rather than to perform the much more intensive engineering studies. Results must be interpreted accordingly, keeping in mind that the actual cost of implementing these controls, as well as the actual reductions that would be achieved, might be significantly different from the generated estimates.

The area source controls analysis for the Baltimore IEMP was more extensive than for the Philadelphia IEMP since it also included a heating source and diesel road vehicles. IEMP generated the cost estimates and pollutant removal efficiencies for heating, dry cleaning, degreasing, miscellaneous industrial solvent usage, and diesel vehicles, while Maryland AMA provided cost estimates and efficiencies for gasoline marketing.

Santa Clara IEMP Control Analysis

The Santa Clara IEMP was undertaken with the assumption that local efforts can complement the work of State and Federal agencies. Therefore, this controllability study provided added emphasis on unique, localized air toxics sources. At the same time, sources of air toxics currently being considered for regulation at the State and Federal level were included in the study to add perspective and provide a point of reference from which to measure the cost-effectiveness of other control strategies.

There are approximately 24 principal source categories (e.g., cement manufacturing, residential wood combustion) of air toxics in the Santa Clara Valley. The following criteria were used to select source types for the controllability study.

- Estimated health risk. Do the pollutants from this source category represent a significant health risk?
- Feasibility of control. Is it technically feasible to control emissions in this source category?
- Data availability. Is there easily accessible information on costs and efficiencies of control measures?

Often, regulatory development programs examine several control options for one source type. The most stringent control option that is economically achievable is chosen as the preferred option. Rather than compare several options for a given source type, the Santa Clara IEMP chose for evaluation what appeared to be the most cost-effective control strategy. In general, only one or two control methodologies were thus considered per source type. In this way, numerous source types were considered in the analysis and compared with each other. Where a risk reduction strategy appears desirable for a particular source type, more in-depth control studies should be performed in which various control options might be considered.

Where possible, the capital and annualized costs of the control strategies were evaluated. Annualized costs consist of annual costs and capital recovery (depreciation and interest on capital). The estimated reductions in excess cancer incidence were also estimated for each control strategy. Generally, the reductions in the incidence rate were calculated from average risk values (i.e., the number of excess cancer cases divided by the overall study population). For a few select source categories, the reduction in risk for a hypothetical maximum exposed individual (MEI) was evaluated.

Cost-effectiveness was measured in terms of the dollars per reduced cancer case. More specifically, cost-effectiveness was calculated by dividing the net present worth (NPW) of the control strategy by the estimated number of cancer cases reduced. The NPW provides an indication of how much would theoretically have to be set aside in the present to provide for the future services of the control option. A time period of 30 years was used; this is the maximum estimated lifetime of the control strategies considered.

Table 6-3 summarizes the results of the cost-effectiveness analysis for cancer incidence reduction in the Santa Clara IEMP. Table 6-3 also shows the specific control methods simulated on the source categories and pollutants selected for analysis.

Analysis of Areawide vs. Most Exposed Individual (MEI) Risk Reduction

The primary controllability emphasis in the IEMPs (and 5 City Controllability Study) is the reduction in areawide cancer incidence. Assessing risks to the most exposed individual (MEI) is another way to evaluate the effectiveness of control alternatives. In the management of risks, policymakers are often faced with value judgments between control options that reduce risks to the overall population and alternatives that reduce risk to the most exposed individuals. Analysis that considers both measures of risks adds clarity to the implications of policymakers' decisions and allows fair consideration of the risks and control costs among different exposed groups of people.

Cost-effective controls to reduce risk to the most exposed individual are not necessarily the same as and may be quite different from the cost-effective controls identified to reduce risk to the general population. Though consideration of these two different measures of risk in a cost-effectiveness framework may result in widely differing control solutions, the contrasting quantitative information places into sharper focus some of the values policymakers must weigh before making public health protection decisions.

Table 6-3. Estimated Cost Effectiveness of Santa Clara IEMP Control Options

Source Type	Control Method	Pollutant	Cancer Incidence Reduction (incidence in 30 years)	Net Present Value for 30 Years of service ($)	Cost Effectiveness (million $/ reduced incidence)[b]
Dry Cleaning[c]	Converting perc transfer to perc dry-to-dry	perchloroethylene	0.0019	56,000	29
	Converting perc transfer to CFC 113 dry-to-dry	perchloroethylene	0.00028	130,000	50
	Converting perc dry-to-dry to CFC 113 dry-to-dry	perchloroethylene	0.00044	87,000	200
Solvent Degreasing	Refrigerated freeboard chillers	perchloroethylene trichloroethylene methylene chloride	0.069 0.021 0.12	2,400,000 290,000 470,000	36 14 4.1
Residential Wood Combustion	Fuel efficient wood stoves[d]	(e)	0.9	420,000	0.47
	Burning curtailments	(e)	0.74	N/A	N/A
Hospital Sterilization	Hydrolizing wet scrubbers	ethylene oxide	0.38	1,200,000	3.1
Semiconductor Manufacturing	Thermal incinerators	cellosolve	(f)		(f)
	Catalytic incinerators	cellosolve	(f)		(f)
Mobile Sources	Oxygen sensor durability[g]	benzene and organic particulate	0.09	0[h]	0
	Modifier certified new vehicle registration[i]	benzene and organic particulate	0.034	0[h]	0
	Mass transit improvements	benzene and organic particulate	0.21	$1,800 \times 10^6$	8,800
	Catalyst retrofit for heavy-duty gasoline vehicles	benzene and organic particulate	2.3	170,000,000	76
	I/M for heavy-duty gasoline vehicles	benzene and organic particulate	0.31	6,600,000	22

[a] The term "average" refers to the overall risk calculated for the entire county population. Because of significant uncertainties in the underlying data and assumptions, these estimates of risk are only rough approximations of actual risk. They are based on conservative estimates of exposure and potency and are more likely to overestimate risks than to underestimate them.

[b] Cost-effectiveness = (net present value of 30 years of service/cancer incidence reduced in 30 years).

[c] The net present value for the dry cleaning control costs are based on one machine.

[d] Control method involves replacing older stoves with newer, less polluting stoves.

[e] Wood stove emissions include benzene, organic particulate, and toxic inorganics. Emissions of these pollutants are all reduced simultaneously, so that the cost of removal and incidence reduction includes all three pollutant categories.

[f] Cellosolve is noncarcinogenic, but believed to cause birth defects. Calculating the reduction in cancer incidence is not appropriate.

[g] Adopted by ARB in 4/85.

[h] Cost of control was attributed by ARB to criteria pollutant reduction.

[i] Adopted by ARB in 12/85.

CAUTION: All emission and risk reductions and cost estimates in this table are shown only for example purposes, and are not intended to apply in other situations and locales.

Unfortunately, population-averaged cost-effectiveness cannot be used in determining MEI cost-effectiveness. If the size of the general population is large and the risk to them low, the average cost-effectiveness of control will be poor regardless of the risk to the maximum exposed individual. In addition, a single, meaningful cost-effectiveness value for the MEI is impossible to calculate without an estimate of the size of the affected population. Thus, several IEMPs computed the total annualized cost of control and cost per pound of pollutant for removal along with the risk reduction to the MEI. Together, these variables were felt to offer qualitative insight into the relative merits of the various controls.

Table 6-4 shows the results of the MEI analysis in the Santa Clara IEMP.

6.6 Insights on Control Strategy Simulation and Evaluation

Perhaps most important, the study manager should be well aware of the limitations of the controllability analysis and should not use any conclusions to support actions unwarranted by the accuracy of the underlying data base or the validity of the assumptions made. All of the studies reviewed were described as screening or scoping studies, and were not intended to predict absolute reductions in risk associated with any particular control strategy. Rather, they were intended to be used in a relative sense only to begin to develop priorities.

Case-by-case emission and control projections at specific facilities are generally more accurate than more general types of projections based on surrogate measures of growth and plant retirement. The latter type of analysis, used in the 5 City Controllability Study, gives only a very broad sense of control potential associated with certain groups of control measures, and may not be appropriately applied to determine control potential and feasibility in a particular locale.

The study manager should plan at the outset of the controllability study just what measures of risk reduction need to be quantified. Generally, the studies reviewed focused on areawide cancer incidence reductions; moreover, it was the easiest to develop cost-

Table 6-4. Estimated Cost Effectiveness of Santa Clara IEMP MEI Control Options

Source Type	Control Method	Pollutant	Emission Reduction (lb/yr)	Capital Cost of Control ($)	Annualized Cost of Control[b] ($)	Cost of Removal[c] ($/lb)	Estimated Baseline Risk	Estimated Risk after Control[b]
Dry Cleaning	Converting perc transfer to perc dry-to-dry	perchloroethylene	16,000	140,000	15,000	1.1	1.2×10^{-5}	2×10^{-6}
	Converting perc transfer to CFC-113 dry-to-dry	perchloroethylene	20,000	150,000	34,000	1.9	1.2×10^{-5}	0
	Converting perc dry-to-dry to CFC 113 dry-to-dry	perchloroethylene	20,000	37,000	9,800	7.4	1.2×10^{-5}	0
Decorative Chrome Plating	Polypropylene balls	chromium[d]	1.5	100	240	170	1.4×10^{-6}	5.8×10^{-7}
Hard Chrome Plating	Packed bed scrubber	chromium[d]	33	29,000	15,000	450	1.1×10^{-5}	1.1×10^{-7}
Degreasing	Refrigerated freeboard chillers	1,1,1-trichloroethane methylene chloride	28,000 4,800	69,000 10,000	18,000 2,800	0.64 0.59	(d) 3.0×10^{-5}	(d) 2.1×10^{-5}
Hospital Sterilization	Hydrolizing wet scrubbers	ethylene oxide	1,700	36,000	9,500	5.5	2.0×10^{-4}	3.2×10^{-5}
Cooling Towers[e]	Switching additives	chromium[d]	2.7	(f)	(f)	(f)		

[a] MEI = maximum exposed individual

[b] Annualized costs = annual operating costs plus capital recovery

[c] Cost of removal = annualized costs/emission reduction lb/yr.

[d] Chromium emissions from chrome plating and cooling towers are all in the plus-six oxidation state.

[e] This analysis includes only the largest tower.

[f] Due to inconsistent vendor information and customized tower additives, we were unable to estimate costs with any accuracy.

CAUTION: All emission and risk reductions and cost estimates in this table are shown only for example purposes, and are not intended to apply in other situations and locales.

effectiveness data for these reductions. While MEI reductions can also readily be projected, they are more difficult to evaluate from a cost-effectiveness standpoint.

None of the studies documented how, if at all, the resulting controllability conclusions would be used in the risk management process, or broached the topic of defining acceptable public risk.

CHAPTER 7

COMPUTERIZED DATA HANDLING

The following topics are presented in this chapter:

- Data handling considerations in urban air toxics studies
- Data handling aspects of emission inventory/dispersion modeling studies
- Data handling aspects of ambient air monitoring studies
- Insights on computerized data handling

7.1 Data Handling Considerations in Urban Air Toxics Studies

The development of exposure and risk estimates in multi-source, multi-pollutant assessments involves extensive data handling. Because exposure and risk estimates are made, both individually and collectively, for many pollutants and receptors across broad geographical areas, computerized data handling is a virtual necessity in most cases. Hence, the study manager should consider the data handling aspects of carrying out an urban air toxics study at the outset of his/her study as part of the overall study protocol. The development of specialized data handling software is expensive and time consuming and can potentially be avoided if existing software can be utilized. Many of the studies reviewed in this report adapted data handling capabilities already in existence rather than developing new capabilities.

Study Type

As discussed earlier, most urban air toxics studies have involved either emission inventory/dispersion modeling studies or ambient air monitoring studies, albeit with some hybridization common. Data handling is more complex for the former study type because one has to work with emission inventory data and dispersion models in order to predict ambient air concentrations of air toxics, whereas these concentrations are measured directly in the latter type of study. In both study types, ambient air concentrations must be applied to population data, usually at some level of spatial disaggregation, to estimate population-averaged exposures. These exposures are, in turn, adjusted by potency values to estimate cancer risks or other health effects.

Computer Accessibility

An obvious consideration is whether the study has access to a mainframe computer. Most State and local agencies can access EPA's mainframe computers at Research Triangle Park, North Carolina through remote terminals. In some cases, these agencies also have access to their own States' computer facilities. EPA maintains an IBM 3090 computer and several DEC VAX computers. EPA's Univac computer, used in several of the studies reviewed, is no longer on line as of October 1988. Many of the dispersion, exposure, and risk assessment models discussed earlier in this report reside on EPA's mainframe computers.

Most agencies now have personal computers, and many agency personal untrained in mainframe languages prefer to operate in the PC environment. Some of the operations involved in urban air toxics assessments, such as dispersion modeling, cannot realistically be carried out using today's PCs, but some operations are possible, including inventory compilation, exposure and risk estimation, and control strategy evaluation. The extent to which these latter operations can be accomplished on PCs will depend on the number of pollutants, sources of pollution, and the spatial resolution reflected in the analysis. Regardless of the extent of mainframe involvement in "number crunching," PCs can be useful for analyzing summary data sets created by the mainframe and for tailoring special reports and graphics. Hence, PC data handling should be considered to complement mainframe analyses in many facets of a study.

PCs are certainly capable of storing air quality monitoring data from one or several sites. Data summaries can be produced using spreadsheets or data base management programs commonly available for most PCs.

7.2 Data Handling Aspects of Emission Inventory/Dispersion Modeling Studies

PIPQUIC

Many of the studies reviewed herein (the Philadelphia, Baltimore, Santa Clara, and Kanawha Valley IEMPS; the Southeast Chicago Study; the 35 County Study within the Six Months Study) used a computerized data handling system called PIPQUIC to store their emissions data and develop estimates of exposure and risk. PIPQUIC was developed for the IEMPs and resides on EPA's IMB 3090 at Research Triangle Park, North Carolina. It is accessible to account holders through any dialup terminal, including desktop PCs. Various terminal emulator software programs may be used for accessing PIPQUIC. (PIPQUIC, 1989)

PIPQUIC offers the user a "tool kit" to produce bar charts, tables, printouts, pie charts, area maps, 3-D maps, contour maps, and spreadsheets. PIPQUIC creates charts, maps, tables, and the like, using air toxics emissions data (or, as a default, data generated by PIPQUIC by applying species factors to NEDS data). PIPQUIC executes two EPA models—ISCLT and CDM—using gridded emissions data and appropriate meteorological data for the area. Optionally, the user can run his/her own models of choice in lieu of ISC and CDM. In either case, the model-predicted ambient concentrations are then coupled with population and cancer potency data to estimate individual cancer risks and aggregate incidence. PIPQUIC stores emissions, modeled concentrations, risk, and incidence data for efficient retrieval and analysis to create tables, charts, and maps.

A discussion of specific PIPQUIC tools is useful as these tools were designed to assist study area managers in answering key questions regarding urban air toxics risks.

Illustrations of example outputs from PIPQUIC are included to show the reader the various ways the data are handled to provide useful summary graphics.

PIPQUIC's Tool 450 creates a broad range of study maps, and allows the user to overlay point sources, modeling grids, and area source emissions data on these maps. Figures 7-1 and 7-2 show examples of several study area maps produced by PIPQUIC.

Tool 440 allows the user to rank order his/her source and emissions data in many ways via bar charts, cross-tabulations, pie charts and printouts. The user can select one or more pollutants, facilities, counties, industries, source types or estimation methods, and can also define variables. Whole values should appear as separate pages, rows, or columns. Figure 7-3 and Table 7-1, respectively, show example bar charts and cross-tabulations created by Tool 440.

Figure 7-1. Example of PIPQUIC Output for Southeast Chicago Study Area

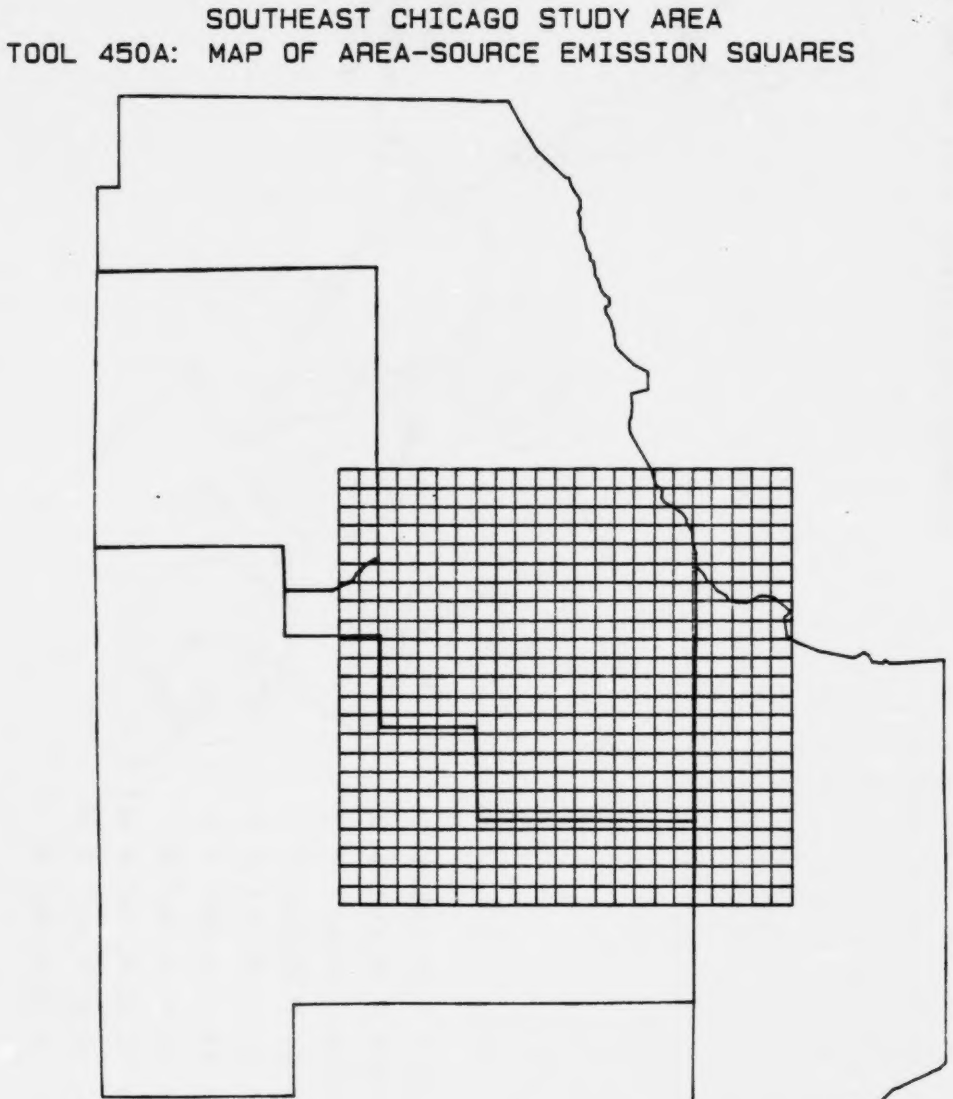

SOUTHEAST CHICAGO STUDY AREA
TOOL 450A: MAP OF AREA-SOURCE EMISSION SQUARES

Figure 7-2. Example of PIPQUIC Output for Staten Island

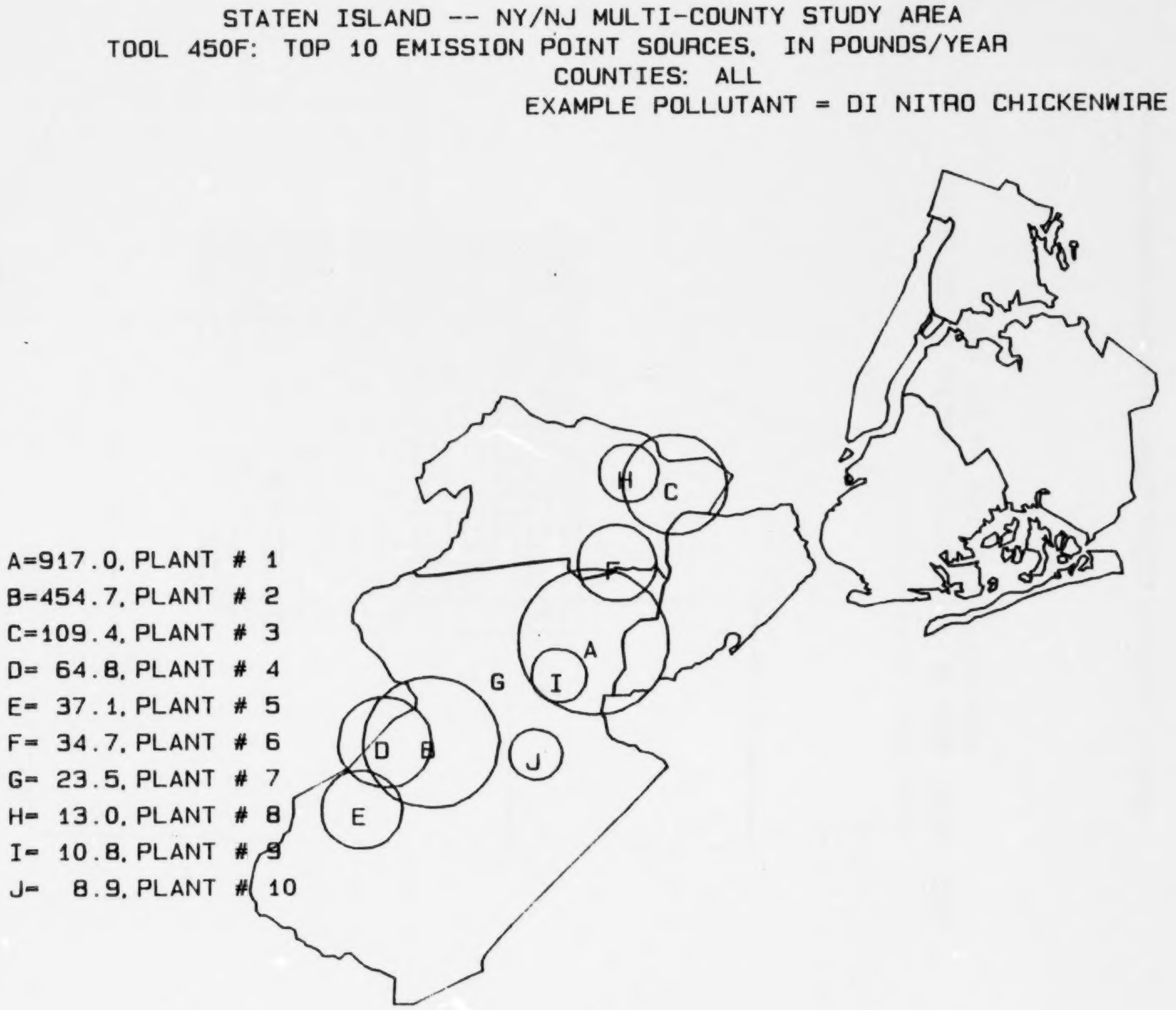

STATEN ISLAND -- NY/NJ MULTI-COUNTY STUDY AREA
TOOL 450F: TOP 10 EMISSION POINT SOURCES, IN POUNDS/YEAR
COUNTIES: ALL
EXAMPLE POLLUTANT = DI NITRO CHICKENWIRE
A=917.0, PLANT # 1
B=454.7, PLANT # 2
C=109.4, PLANT # 3
D= 64.8, PLANT # 4
E= 37.1, PLANT # 5
F= 34.7, PLANT # 6
G= 23.5, PLANT # 7
H= 13.0, PLANT # 8
I= 10.8, PLANT # 9
J= 8.9, PLANT # 10
H
C
F
A
I
G
D
B
J
E

Figure 7-3. Example of PIPQUIC Output for Urban Study Area

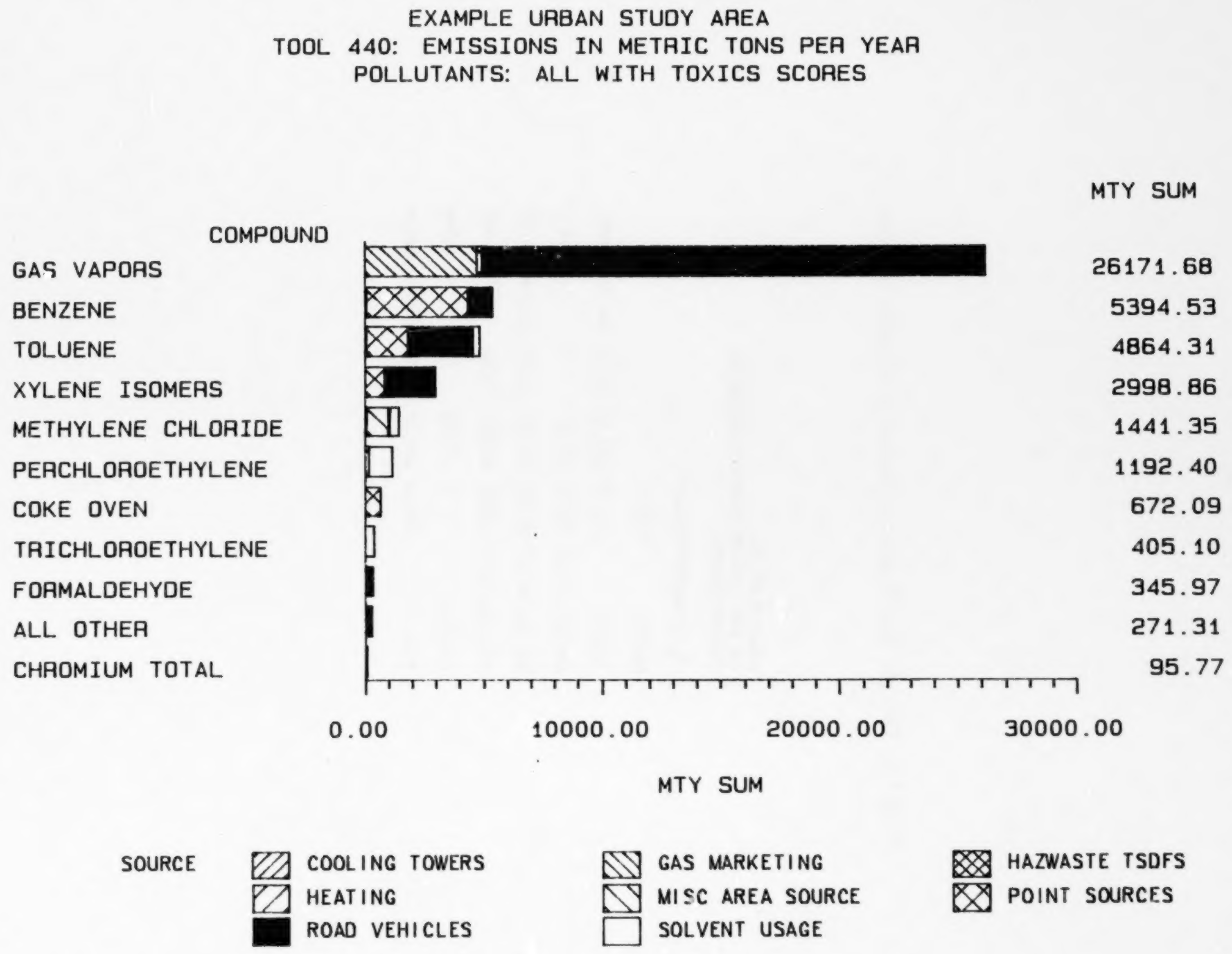
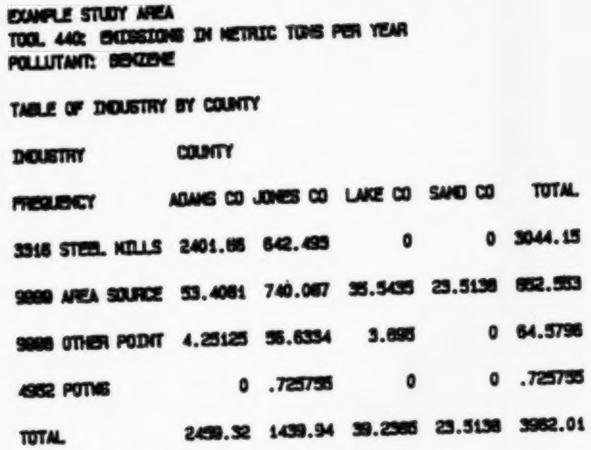


Table 7-1. Example Cross-Tabulation Created by PIPQUIC Tool 440



```
EXAMPLE STUDY AREA
TOOL 440: EMISSIONS IN METRIC TONS PER YEAR
POLLUTANT: BENZENE

TABLE OF INDUSTRY BY COUNTY

INDUSTRY         COUNTY

FREQUENCY        ADAMS CO JONES CO  LAKE CO  SAND CO    TOTAL

3316 STEEL MILLS  2401.66  642.495        0        0  3044.15

9999 AREA SOURCE  53.4081  740.087  35.5435  23.5138  852.553

9998 OTHER POINT  4.25125  56.6334    3.695        0  64.5796

4952 POTWS              0  .725755        0        0  .725755

TOTAL             2459.32  1439.94  39.2385  23.5138  3962.01
```

PIPQUIC's Tool 453 allows the user to pinpoint the sites of maximum concentrations, individual risk, and aggregate cancer incidence and to assess the impact of each pollutant and source at any receptor within the study area. Output options include bar charts, 3-D maps, contour maps, charts ranking receptors, charts ranking sources culpable at the point of maximum individual exposure (or at any other point), and tables of concentration and risk. Tool 453 is perhaps PIPQUIC's most powerful analysis tool. Figures 7-4 through 7-6 and Tables 7-2 and 7-3 show example graphics created by Tool 453, which characterize emissions data and associated exposure and risks in effective ways to help understand the nature and magnitude of the air toxics problem in a given area.

PIPQUIC enables the user to download maps, graphs, and the like, to his/her desktop computer and then to re-create and edit them without having to re-enter PIPQUIC. Using various inexpensive PC software packages, the user can edit titles, change colors, assemble video presentations, route to color plotters, or convert downloaded files into spreadsheet or graphics files for detailed editing. As a specific example, PIPQUIC's Tool 123 supports downloading of source, emissions, and aggregate incidence data for creating LOTUS 1-2-3® spreadsheets to evaluate control scenario effectiveness.

Other Data Handling Systems

Several other emission inventory/dispersion modeling studies developed their own data handling capabilities. The 5 City Controllability Study developed input/output software around EPA's HEM/SHEAR model, whereas the South Coast Study did likewise around its modified version of HEM called SCREAM. The 5 City Study also produced a series of files containing regulatory projection information to run a Regulatory Impact Model (RIM). RIM allows the user to project future emissions and cancer incidence by simulating various hypothetical control strategies. HEM/SHEAR and SCREAM are mainframe models, whereas RIM is a PC-based model. As of this writing, HEM/SHEAR is being updated by EPA and converted to a VAX computer at Research Triangle Park, North Carolina. HEM/SHEAR will be accessible to users having an account on the EPA VAX

Figure 7-4 Example PIPQUIC Output - Incidence

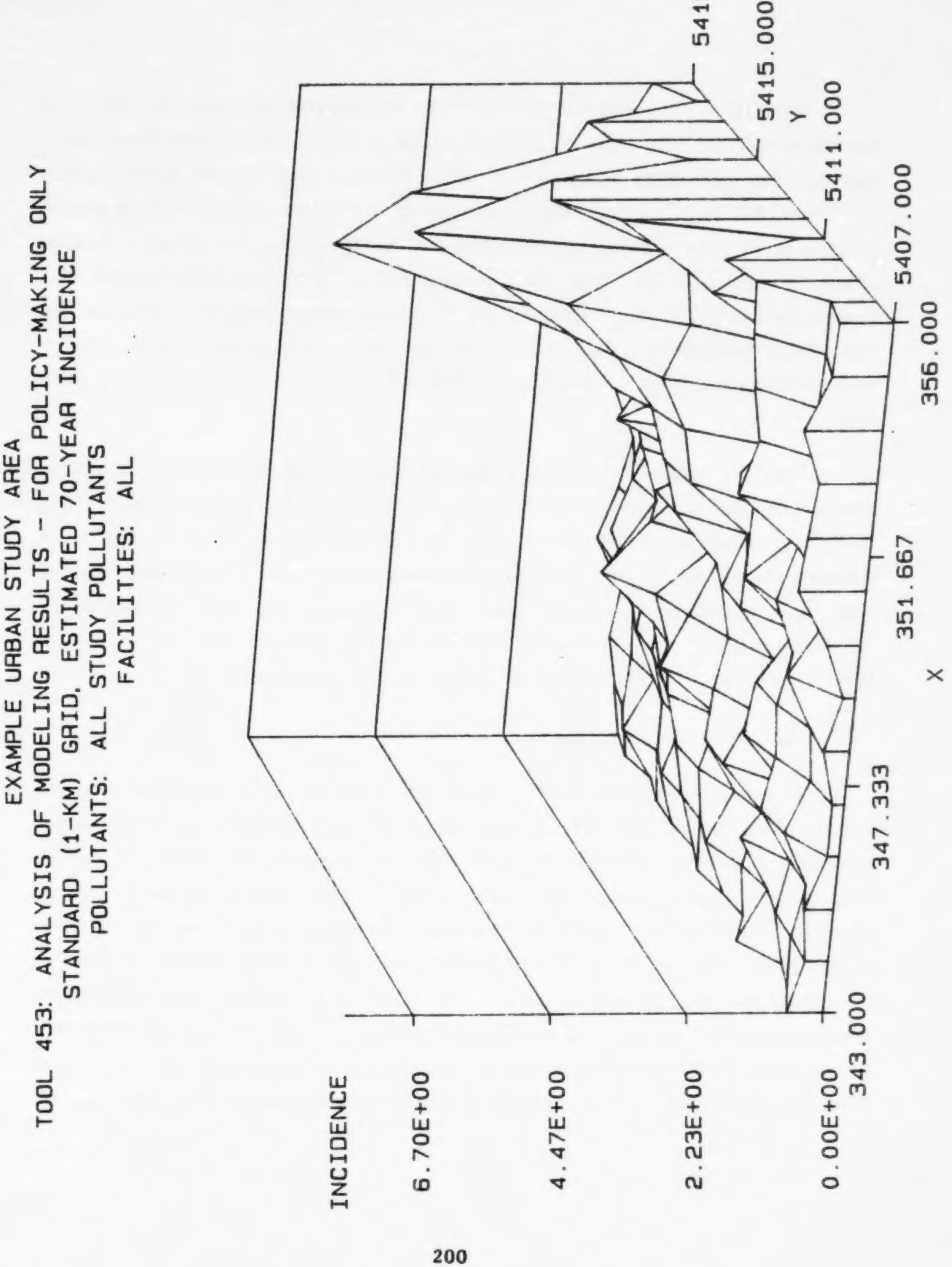


Figure 7-5. Example of PIPQUIC Output — Benzene Contours

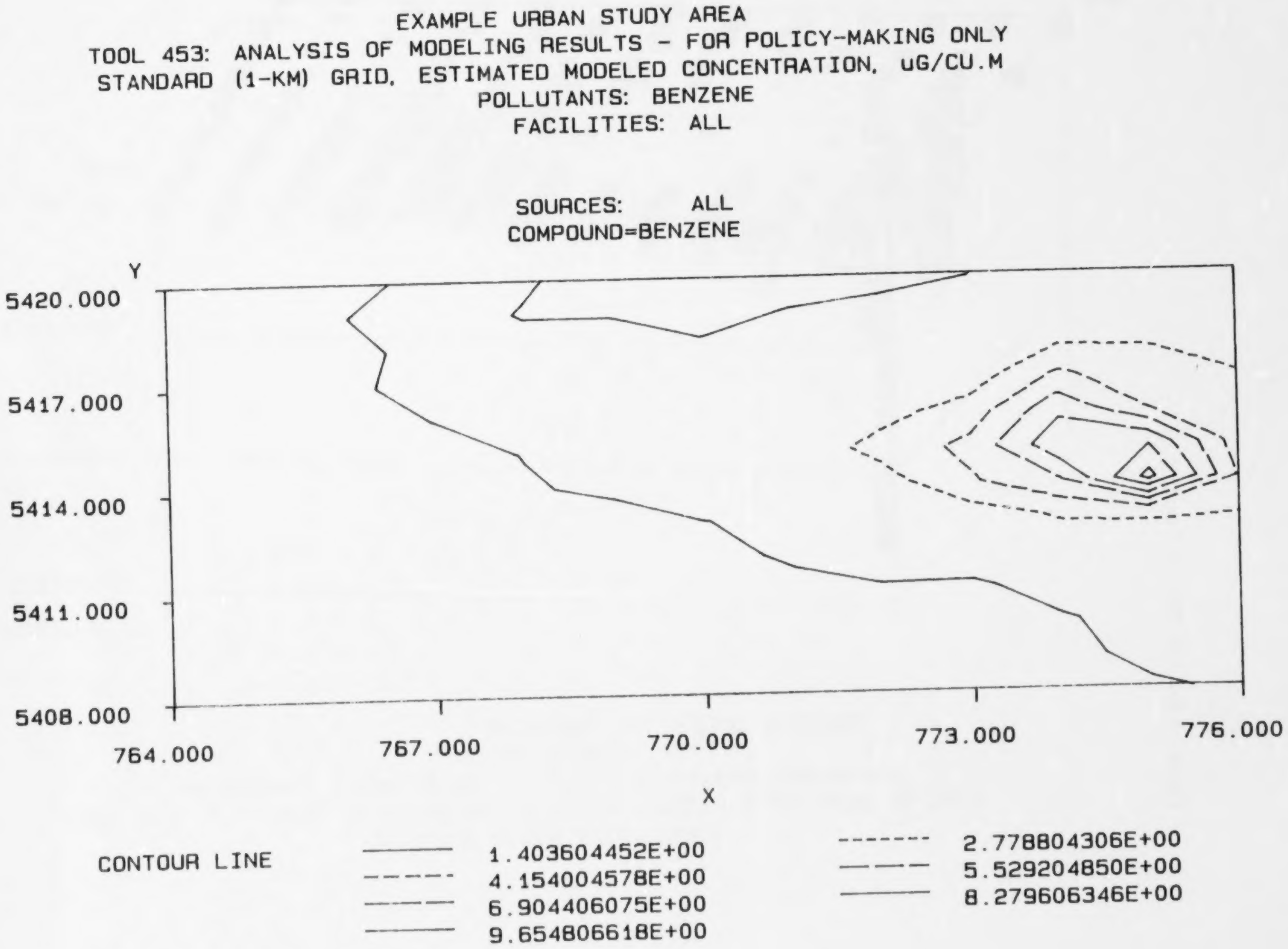


Figure 7-6 Example PIPQUIC Output

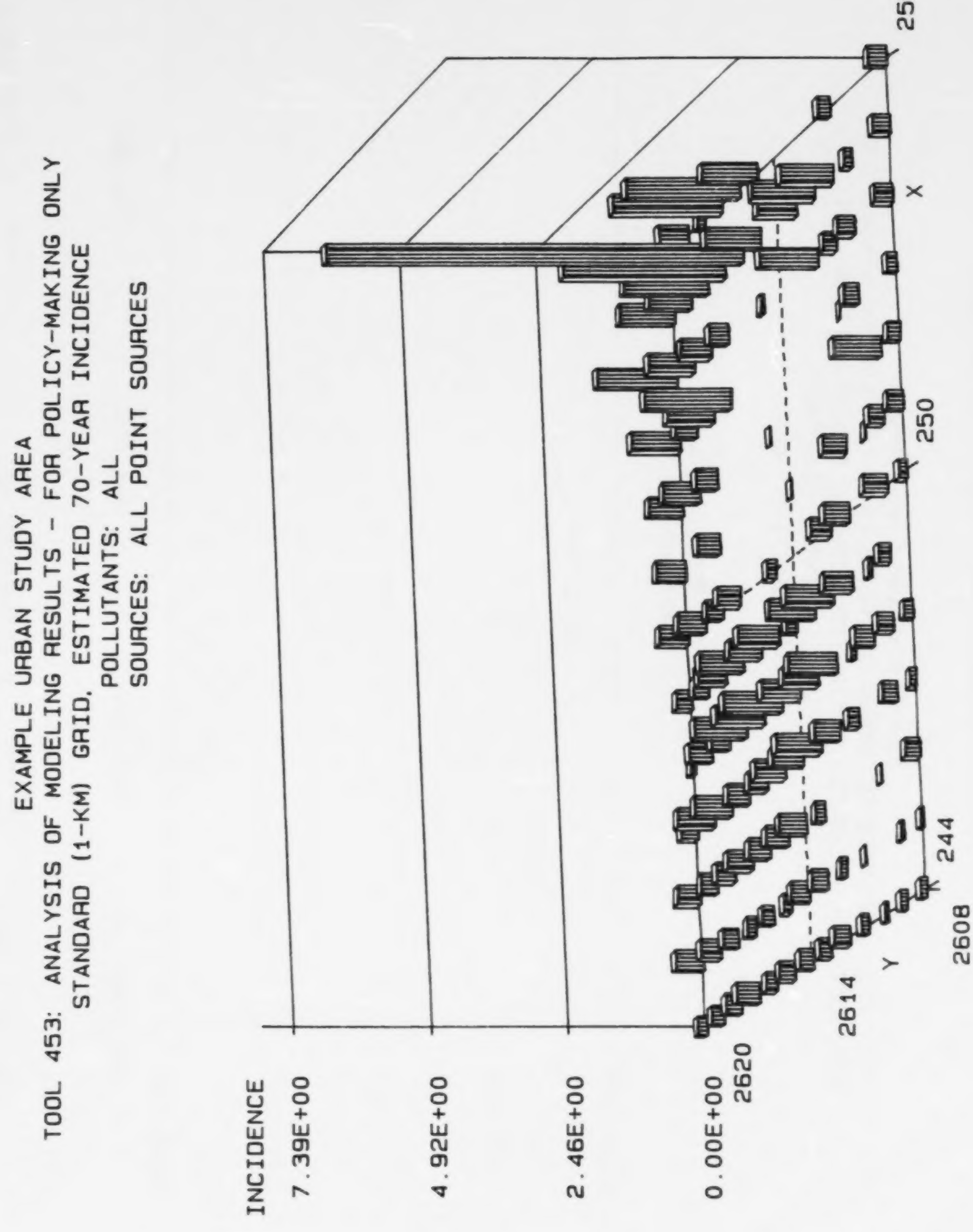


Table 7-2. Example Cross-Tabulation Created by PIPQUIC Tool 453

```
XYZ STUDY AREA
TOOL 453: ANALYSIS OF MODELING RESULTS - FOR POLICY-MAKING ONLY
STANDARD (1-KM) GRID, ESTIMATED 70-YEAR INCIDENCE
POLLUTANTS: ALL C.A.G. POLLUTANTS
FACILITIES: ALL
RECEPTORS:  4616.00 454.00
SOURCES:    ALL

TABLE OF COMPOUND BY SOURCE

COMPOUND         SOURCE

FREQUENCY        POINT    ROAD    MISC  COOLING   GAS SOLVENT HEATING  TOTAL
               SOURCES VEHICLES   AREA   TOWERS  MKTG   USAGE

COKE OVEN       4.4075                                               4.4075
ARSENIC         0.8915                                               0.8915
BENZENE         0.3055  0.0832 0.00011         0.0022 0.00045        0.3916
CADMIUM         0.3694  0.0011                                       0.3705
1,3-BUTADIENE           0.2394                                       0.2394
GAS VAPORS              0.1459                 0.0330                0.1790
CHROMIUM HEXAVAL               0.05016 0.04104                0.0114 0.1026
METHYLENE CHLORIDE             0.03648                0.01254        0.0490
FORMALDEHYDE    0.0011  0.0285 0.00684                        0.0068 0.0433
ETHYLENE OXIDE                 0.01026                               0.0103
CHLOROFORM                     0.00684                               0.0068
PERCHLOROETHYLENE                                     0.00570        0.0057
TRICHLOROETHYLENE                                     0.00342        0.0034

TOTAL           5.9751  0.4982 0.11070 0.04100 0.0353 0.02210 0.0182 6.7007
```

Table 7-3 Example PIPQUIC Output

```
EXAMPLE URBAN STUDY AREA
TOOL 453:  ANALYSIS OF MODELING RESULTS - STANDARD GRID
ESTIMATED INDIVIDUAL RISK - POLLUTANTS:  CADMIUM BENZENE ARSENIC
FACILITIES:  ALL

RECEPTOR UTM COORDINATES                         FREQ    IND RISK
613.00 55.00   |AAAAAAAAAAAAAAAAABCCCCCC 246 1.21194E-03
616.00 55.00   |AAAAAAAAAABCCCC          246 7.09339E-04
615.00 54.00   |AAAAAAAABCCC             246 6.24182E-04
616.00 54.00   |AAAAAAABCCC              246 5.63863E-04
614.00 55.00   |AAAAAABBCC               246 4.91408E-04
614.00 54.00   |AAABC                    246 2.62701E-04
617.00 54.00   |AAABC                    246 2.50146E-04
616.00 56.00   |AAABC                    246 2.43519E-04
618.00 55.00   |AAAC                     246 2.42866E-04
613.00 56.00   |AAAC                     246 2.18210E-04
612.00 48.00   |AAC                      246 1.69424E-04
616.00 52.00   |A                        246 8.86776E-05
ETC.
                ----------------+---------
                        8.000000000E-04  70 YR INDIVIDUAL RISK
SYMBOL  COMPOUND      SYMBOL  COMPOUND      SYMBOL  COMPOUND
  A     ARSENIC         B     BENZENE         C     CADMIUM
```

computer, and a user's guide, reflecting the updates made to the model, will be forthcoming. A user'sguide for the current version of HEM/SHEAR model is presently available. (EPA, 1986) The SCREAM model is applicable only to the Los Angeles geographical area.

Cost Saving Techniques

Normalized Modeling—As mentioned in Chapter 4, the practice of normalized modeling will minimize the number of dispersion model runs necessary in an urban air assessment. This practice will commensurately save on data handling expense. Normalized modeling was done in most of the emission inventory/dispersion modeling studies reviewed in this report.

For example, instead of running HEM/SHEAR separately for each pollutant and each emission projection, the 5 City Controllability Study used a single run, assuming 100 tons per year of pollutant was emitted from each point source. For each point source, the output of SHEAR was saved in an intermediate file containing the cumulative population exposure (microgram-persons/cubic meter-year) for each modeled point. These cumulative values were thus used to estimate population exposures to individual pollutants by multiplying them by the ratio of actual-to-modeled emissions.

Modeling Small Point Sources as Area Sources—Because the cost and execution time of modeling point sources is much greater than for modeling area sources, the 5 City Controllability Study opted to treat small point sources as area sources if they emitted below a particular cutoff level. This minimum cutoff level varied by pollutant and was determined to some extent to be a function of the number of small sources that emitted a particular pollutant within the study area, as well as by the toxicity of that pollutant.

7.3 Data Handling Aspects of Ambient Air Monitoring Studies

Limited information is available on data handling specifics in the ambient air monitoring studies reviewed herein. Two of the studies for which some information is available—the Staten Island/Northern New Jersey Study and the Urban Air Toxics Monitoring Program—are only in the data collection phases at present and have not yet completed any exposure or risk assessment. Both of these studies are using Lotus 1-2-3 spreadsheets to store the raw data and to develop summary reports. Table 7-4 is an example of a quarterly report generated in the Staten Island/Northern New Jersey Study.

7.4 Insights on Computerized Data Handling

Data handling in urban air assessment studies involving dispersion modeling, exposure/risk assessment, and control scenario evaluation can become quite complex and should be carefully considered when developing the study protocol. The agency may want to consider contractual assistance in this area.

The study area manager should consider using available mainframe software for conducting his/her urban study. EPA maintains various dispersion models as well as exposure/risk models that can perform many of the core data handling functions necessary in an urban air toxics assessment.

Many data handling functions can be performed efficiently and more readily on a PC than on a mainframe computer. Specifically, the preparation of emissions data and other data needed to run dispersion models can be done on a PC in cases where extraordinarily large data bases are not involved. Additionally, the outputs can effectively be downloaded to desktop PCs for editing and analysis, and for the creation of summaries and graphics.

Data handling complexity and costs can be reduced by normalized modeling and by modeling small point sources as area sources. Both techniques result in fewer point source modeling runs. The reader is cautioned, however, that there are drawbacks to both techniques. Normalized modeling assumes a linear relationship between emission changes and model-predicted concentrations. This assumption may be invalidated if the release specifications change as emissions change (e.g., a control device may alter a plant's stack/exhaust parameters as well as its emissions) and, hence, may need to be carefully examined in detailed assessments. Also, the treatment of small point sources as area sources may change the exposures resulting from those sources, as their emissions will subsequently be "smeared out" over entire grid cells and could be assumed to be emitted

Table 7-4 Example of Ambient Air Data Set Summary

Reduced Data from All Sampling Systems (Quarterly Report)

Agency: College of Staten Island

Pollutant:Chloroform CAS #: 67-66-3 ______________

Quarter Beginning (Month, Year): Jan88

MDL: 0.04 ppb low flow & 0.02 ppb high flow Units: ppb

SAROAD #	Site	Sampling Code	Analytical Code	# of Samples	Arith. Mean	Std. Dev.	1st Max	2nd Max	Min	# > MDL	FC
8	Bayley Seton	Tenax	GC/MS	82	0.039	0.037	0.321	0.125	0.008	81	
3	Eltingville	Tenax	GC/MS	57	0.043	0.023	0.121	0.112	0.011	57	
6	Dongan Hills	Tenax	GC/MS	55	0.046	0.037	0.269	0.111	0.012	55	

at ground level. Such a distribution of emissions from small sources could potentially overemphasize the risks from these small point sources, especially if they are treated as area sources that are subcounty-apportioned by population.

Chapter 7

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CHAPTER 8

EMERGING METHODS: RECEPTOR MODELING AND BIOLOGICAL TESTING

The following topics are presented in this chapter:

Receptor modeling

- Use of receptor modeling in urban assessments
- Pollutants and source categories addressed
- Source signature testing and tracer analysis
- Measured ambient air quality data sets
- Statistical techniques used for source apportionment
- Spatial and temporal representativeness of results
- Comments on the use of receptor modeling

Biological testing

- Approach to use of biological testing in urban air toxics studies
- Comments on the use of biological testing

8.1 The Use of Receptor Modeling in Urban Assessments

Receptor modeling—also called source apportionment—is an evolving science and is not yet widely implemented in urban air toxics assessments. Receptor modeling techniques were originally developed to study sources of particulate matter, and have been used to identify sources of certain toxic metals and extractable organic matter within the particulate catch (Lioy, 1988). Recently, investigators have begun to use receptor modeling to study sources of VOC (O'Shea, 1988; Scheff, 1987), which can yield information on specific gaseous toxics such as benzene. The Denver IEMP will attempt to apportion gaseous VOC if adequate data are acquired during the air monitoring phase of the study. In addition, EPA's Integrated Air Cancer Project (IACP) is using receptor modeling to apportion the mutagenic activity of ambient particulate matter between mobile sources and wood smoke.

The purpose of receptor modeling is to estimate contributions of sources to monitored pollutant concentrations at specific receptor sites. The process employs a variety of statistical techniques to identify the site-specific impacts of pollutant sources, or source categories, on the basis of their emission "signatures" or "fingerprints." Several basic signature types may be employed. One type of signature is the specific mixture of chemical species emitted from a particular source, identifying the ratio or relative concentration of each chemical species to the whole quantity of pollutant in an emission stream. For instance, if one source emits two grams of benzene for every one gram of toluene, its impacts may be distinguished, through statistical analysis, from another source that emits two grams of toluene for every gram of benzene. Another signature type involves the use of unique "tracer" pollutant in sources' emissions. For instance, lead and bromine emissions are associated with mobile source emissions, potassium and iron with wood burning emissions, and so forth. These tracers can be used in conjunction with other source signature information to determine source contributions at a receptor point. Since receptor modeling is dependent on the availability of source/emissions chemical composition data, it can be applied only to those pollutants for which adequate emissions data are available for all sources, or source categories, in the study area emitting those pollutants.

Advantages of receptor modeling are that it can (1) confirm source contributions estimated through air dispersion modeling and (2) provide data on source contributions where air dispersion modeling has not been done, where modeling results are suspect because of terrain or meteorological complexities, or where uncertainties exist regarding atmospheric transformation mechanisms. As receptor modeling techniques improve—particularly as hybrid techniques more comprehensively integrate emissions, dispersion and transport, and measured data—receptor modeling may significantly improve the reliability of policy conclusions about the nature of the urban soup problem.

The disadvantages of receptor modeling are its technical complexity, its data requirements, and the inherent limitations of the monitoring data (and sometimes modeling data) upon which it relies.

Pollutants and Source Categories Addressed

Past receptor modeling analyses have dealt almost exclusively with particulate matter. Little has been done on gaseous pollutants, although theoretically these could be addressed as long as the signatures of sources, or source categories, are sufficiently different to allow for differentiation (Pace, 1987). The current IACP and Denver IEMP studies may further develop the appropriate techniques.

Both the IACP and Denver IEMP studies are using receptor modeling primarily to distinguish between mobile source and wood combustion contributions to ambient particulate concentrations. The IACP Boise study will also include the apportionment of specific gaseous pollutants and pollutant classes. The Denver study will attempt apportionment of gaseous pollutants only if the study's monitoring program develops adequate data on the pollutants of concern. (Stevens, 1987)

At this time, neither of these studies plans to use receptor modeling to distinguish between point and area source categories, although some work may be done with power plants and refineries in the Denver study.

Source Signature Testing and Tracer Analysis

The IACP study is performing source testing for wood combustion and mobile sources to complement the source signature data available in the literature. Similarly, the Denver IEMP will test mobile sources and power plants to compile sufficient signature data for receptor modeling. (Stevens, 1987)

In certain kinds of receptor modeling, tracers are used as unique signatures for certain source categories. The following tracers are commonly used to identify source categories:

Mobile Sources:	Lead, bromine, carbon monoxide
Power Plants:	Sulfur, selenium, arsenic
Wood Burning:	Potassium, iron
Refineries:	Lanthanides
Incinerators:	Zinc

Measured Ambient Air Quality Data Sets

Pollutant coverage and representativeness in time and space of the measured ambient air quality data set are important considerations when drawing conclusions from the receptor modeling analysis, as discussed below:

Diurnal Coverage—The monitoring programs for the IACP and Denver studies are very similar; the Denver study, in fact, was patterned after the IACP approach. Each study relied on 12-hour sampling to separate diurnal trends, although in Denver a combination of 12- and 24-hour samples was collected to reduce costs. In these studies, it is important to be able to resolve the chemical species emitted from residential wood burning (predominantly a nighttime activity) and mobile source emissions (predominantly a daytime activity). Hence, the 12-hour sampling periods each day were extended from 7 a.m. to 7 p.m. and from 7 p.m. to 7 a.m.

Pollutant Coverage—Chapter 2 discussed the pollutant coverage of these programs in general terms. Pollutants collected specifically to support the receptor modeling include sulfates and nitrates, elemental and organic carbon, elemental analysis, and, in the case of IACP, carbon-14 data to help separate wood from fossil fuel combustion sources. CO data are collected at some sites, along with other criteria pollutants and meteorological data, to provide additional input on source contributions. The IACP study also analyzed samples for mutagenicity to apportion biological activity to appropriate source categories.

Seasonal Coverage—The results of a receptor modeling analysis are directly applicable to the monitoring period for the ambient air quality data set. Both the IACP and the Denver study are performing monitoring in distinct seasonal blocks. For the IACP, the major emphasis will be on the winter season. The Denver study will provide a more balanced emphasis between the winter and summer seasons.

The two-season coverage in Denver provides a more representative data set for estimating annual average source-receptor relationships. The inclusion of two seasons in Denver provides more data for estimating source culpability on an annual average basis.

Spatial Coverage—The Denver study has three ambient air quality monitoring sites in the air toxics monitoring network; a fourth (supplemental) site was available during the winter season. The IACP studies have used as many as seven fixed sites to support the receptor modeling analysis, although only two sites were used in the earlier testing in Raleigh, North Carolina and Albuquerque, New Mexico. With only a few sites there is a possibility that the results of source apportionment do not represent averages for the metropolitan area being evaluated.

Statistical Techniques Used to Estimate Apportionment

The statistical techniques used in receptor modeling strive to find the best fit between the measured data at one or several receptors and the source signature data. It is beyond the scope of this report to address these techniques in detail. References such as EPA 1981, 1983, 1985 provide more comprehensive treatment of this subject. The following is a capsule description of these techniques:

Chemical Mass Balance—The CMB method is based on the assumptions that the mass of material deposited on a filter at a receptor site is a linear combination of the mass contributed from each of the sources and that the mass and chemical speciation are conserved from the time of emission to the time it is measured at a receptor site. The measured data and the data on source signatures are used to form a set of simultaneous equations. There are as many equations as there are chemical species being addressed. The best fit for the set of equations is identified.

Factor Analysis--Whereas CMB methods apply knowledge about source characteristics to a single filter data set to derive a source's contribution, multivariate methods such as factor analysis extract information about a source's contribution on the basis of the variability of elements measured on a large number of filters. If two or more chemical components originate from the same source, their variability as a function of time as measured at a receptor site is assumed to be similar.

Multiple Linear Regression Analyses—This analysis provides a means of calculating the mass of emissions from a given source once the tracer species from that source are known.

Hybrid Receptor Modeling—The measured ambient data and the dispersion term between sources and monitoring sites are used as known values, and the emission rate from each source is solved as the unknown.

To date, the IACP has only used multiple linear regression because the studies are reviewing only two source categories—wood combustion and mobile sources—and more complex models are not needed. As more complex airsheds are studied, future IACP studies may include more complex receptor modeling techniques such as chemical mass balance, factor analysis, and hybrid receptor modeling. For example, factor analysis may be used in the IACP to provide groupings of chemical species (both organic and inorganic) that are characteristic of the emissions and transformation products of sources within the airshed under study.

Spatial and Temporal Representativeness of Results

The Denver study plans to add a step in the data interpretation that will evaluate the representativeness of the ambient air quality measured data set for areas in the city beyond the receptor locations and for periods not covered during sampling. Two techniques—dispersion modeling and use of CO data as a surrogate to toxic air pollutants—will be used to extrapolate the receptor modeling results to develop more general conclusions regarding culpability.

Dispersion Modeling—There are clear limitations to the use of dispersion models in Denver because of the city's topography. The complex drainage flows that occur during periods with peak concentrations are relatively difficult to model, considering the available wind data and the cost of performing non-Gaussian modeling to address the complex trajectories. Confidence in modeled exposure estimates during peak days may be particularly low because of these factors. Dispersion modeling will, however, be used as input to the assessment of the spatial representativeness of the monitoring sites and the representativeness of the monitoring periods to typical concentrations. This link will provide some limited input to support the extrapolation of the results.

CO Measured Data as a Surrogate for Toxic Air Pollutants—As already noted, the Denver air toxics monitoring network has four sites; CO coverage is available at these sites as well as at three additional sites located in widely varying settings, including a downtown site, a residential site, and a relatively remote site.

CO should be a valuable indicator of the magnitude of mobile source impacts at each monitoring site because (1) mobile sources are the dominant source category for both CO and air toxics emitted in the Denver area (Machlin, 1986), and (2) it is reasonable to expect that the measured data for CO could be used to estimate the general magnitude of some air toxics for sites where CO data only were available.

8.2 Comments on the Use of Receptor Modeling for Urban Air Toxics Studies

All receptor modeling analyses are limited by the representativeness of the ambient air quality monitoring sites used to support their general conclusions. The primary disadvantage of receptor modeling is that a limited number of locations (monitoring sites) are often used to support general conclusions regarding urban-scale impacts. For example, it could be interpreted from a residential sample obtained during the winter that 30 percent of the fine particulate concentrations in an urban area are from mobile sources and 70 percent are from wood combustion. It is always essential, therefore, to place the receptor modeling results in proper spatial and temporal context. Any extrapolation of the results beyond this point must be supported by a justification of the representativeness of the data.

Expanded use of dispersion modeling in conjunction with receptor modeling analyses provides a means for assessing the representativeness of the measured data in time and space, which should improve the interpretation of the results of a receptor modeling analysis. Since the IACP and previous Denver studies do not perform this interpretive step, receptor modeling could be strengthened in this area.

It is assumed that the ratios of signature pollutants will remain unchanged during transport from the source to the receptor. If significant changes do occur during transport, the apportionment of impacts among sources could be biased.

The measurements need to be precise enough to distinguish among sources; that is, if little difference is discerned among signatures of potential sources, the imprecise nature of the measuring techniques could blur the differences among the sources under review.

Ambient concentrations should be sufficiently above the detection limits as to allow accurate quantification. This could be a limiting factor for many compounds and could be addressed by a complementary dispersion modeling analysis.

The study area in question should not contain other significant sources that are not considered in a receptor modeling analysis.

The IACP methodology recommends that at least 40 daytime and 40 nighttime samples be available to characterize a season (Stevens, 1987), which can be resource intensive. Similarly, research into hybrid receptor modeling techniques has also indicated that relatively detailed temporal coverage is needed in order for receptor modeling to be effective. (Draxler, 1987) There is some leeway in terms of the minimum number of samples that is considered reasonable for attempting source apportionment (Stevens, 1987); however, any study considering this approach should devote adequate resources to the ambient monitoring and possibly to source testing to assure its effectiveness.

8.3 The Use of Biological Testing in Urban Air Toxics Studies

Testing of toxic air pollutants on living organisms has attracted increased interest over the past several years, but this approach is still basically experimental. By exposing microorganism, whole animals, or selected plant species to potential genotoxicants, biological monitoring provides a method of assessing the potential biological effects of previously untested chemicals or chemical mixtures. Generally, bioassay techniques are limited to assessing direct mutagenic effects; by comparing the number of revertants to known mutagens, bioassaying can be used as a relative indicator of mutagenic activity. These tests are not used to identify tumor promoters or other nonmutagenic effects in environmental monitoring applications.

The advantage of biological testing as part of multi-pollutant, multi-facility studies is that such testing can address interactions among pollutants in a complex mixture, thereby reducing study uncertainties and characterizing sources and fractionated samples of ambient air in terms of their relative mutagenicity.

Biological testing has been conducted primarily in the IACP studies, although some testing has also been done in California, Connecticut, and New Jersey. Current testing has been carried out in research programs rather than in support of regulatory programs. For this reason, this chapter will provide only a brief overview of biological testing in order to introduce this technique as an approach to be considered in future applications. Providing details on biological testing is beyond the scope of this report. Details are available in (Claxton, 1987) and (Lioy, 1988).

At present, biological testing has a number of practical and theoretical disadvantages. Because biological testing is still experimental, there are a number of purely practical problems in attempting to apply it within current field-oriented air toxics surveys. For instance, chemicals such as acetone and toluene/ethanol, which are used to extract samples, can produce artifacts that affect the estimates of mutagenicity. The relatively small mass available from typical ambient air samples is another important factor: it limits the degree of fractionation and the subsequent chemical and biological testing that can be performed. Similar practical problems exist with recovery rates of pollutants from collection media and contamination and/or loss of material in the collection medium.

The theoretical development of the technique is also limited. For instance, experience with gas phase bioassay techniques is still limited (Claxton, 1985; Hughes, 1987). Field studies to date generally do not cover gas phase pollutants in a comprehensive sense because of limitations in the sensitivity of the techniques. Existing studies to address gas phase pollutants have only been carried out in the laboratory, at higher than ambient conditions. At a more basic level, additional research is needed on evaluating relationships between mutagenic activity and human or animal data on carcinogenicity or other health impacts.

Approach

The IACP relies on short-term bioassay tests primarily because they require less massive quantities of the sample, are rapid enough to guide the chemical speciation work, and correlate well with known carcinogenesis studies (Claxton, 1987). Two types of biological testing are used in the IACP:

1. Ames testing is used for large samples, i.e., when more than 10 mg of organic material are available. Ames test procedures allow for the calculation of dose-response relationships, mutagenicity slopes, and so forth. In the IACP studies, Ames tests are used to compare combustion-impacted ambient air samples to background (clean) ambient air samples, indoor ambient air samples, and the combustion source emissions. The IACP is currently examining wood smoke-affected ambient air sites.

2. For relatively small samples, high performance liquid chromatography (HPLC)-coupled liquid pre-incubation assay is used to provide a bioassay "fingerprint" of the sample. Each HPLC fraction is characterized as to its mutagenicity. If the mutagenic fractions are not highly toxic, an indication of their relative mutagenicity is available as revertants per plate per fraction recovered.

8.4 Comments on the Use of Biological Testing in Air Toxics Studies

Relative Importance of Gas Phase Pollutants

Perhaps the most significant finding thus far is that the mutagenicity of the gas phase pollutants, after aging and transformation, can be much greater than either the "fresh" gaseous pollutants or the particulate matter in urban air. This conclusion is tentative, however, and more research is needed to corroborate initial indications.

Nevertheless, it suggests that if biological testing were used as a comprehensive screening technique, future applications would have to address gaseous aged pollutants as well as the more routinely studied particulate matter.

Indoor/Outdoor Differences

In another finding of the IACP study, particulate mutagenicity levels inside a subset of residences were found to be lower than those immediately outside the home. The basis of this finding, as well as the gas versus particle phase issue mentioned above, will undoubtedly be subjects of future research.

Use of Biological Testing to Set Priorities

Biological testing could provide a broad check of genotoxic risks among different metropolitan areas, though variations resulting from the use of different strains of the same species, or varying laboratory conditions, could limit the accuracy of these comparisons. Overall, however, biological testing could become a useful means of identifying urban or industrial areas where a more detailed review of source-receptor relationships for specific pollutants is warranted.

Chapter 8

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APPENDIX A

GLOSSARY OF TERMS

Acute exposure:	One or a series of short-term exposures generally less than 24 hours.
Additive risk/incidence:	Risk/incidence due to the interaction of two or more chemicals in which the combined health effect is equal to the sum of the effect of each chemical alone.
Aggregate incidence:	In an urban air toxics context, an areawide, additive incidence. This term sometimes refers to areawide incidence or additive incidence only, so one should pay attention to the contextual use for the proper meaning.
Ambient (air) monitoring:	The collection of ambient air samples and the analysis thereof for air pollutant concentrations.
Annual incidence:	Lifetime cancer incidence adjusted to a yearly basis, typically by dividing lifetime incidence by 70.
Area source:	Any source too small and/or numerous to consider individually as a point source in an emissions inventory.
Areawide average individual risk:	Average individual risk to everyone in an area (but not necessarily the actual risk to anyone). May be computed by dividing lifetime aggregate incidence by the population within the area.
Areawide incidence:	Incidence over a broad area, such as a city or county, rather than at a particular location, such as an individual grid cell.
ATM:	Atmospheric Transport Model. A Gaussian point source dispersion model used in GAMS (before incorporation of ISCLT for this purpose).

Averaging period:	The length of time over which concentrations are averaged, such as 1 hour, 8 hours, or 24 hours.
B(a)P:	Benzo(a)pyrene. One of a group of compounds called polycyclic organic matter (POM). B(a)P is sometimes used as a surrogate for all POM in computing emissions, exposures, and risks.
Background:	A term used in dispersion modeling representing the contribution to ambient concentrations from sources not specifically modeled in the analysis, including natural and man-made sources.
BACT:	Best available control technology.
BG/ED:	Block group/enumeration district, as designated by the Bureau of Census. A block group is an area representing a combination of contiguous blocks having an average population of about 1,100. An enumeration district is an area containing an average of about 800 people and is designated when block groups are not defined.
Bioassay:	A test in living organisms, e.g., a test for carcinogenicity in laboratory animals, generally rats and mice, which includes a near-lifelong exposure to the agent under test.
Biological testing:	See Bioassay.
Box model:	A simplified modeling technique that assumes uniform emissions within an urban area and uniformly mixed concentrations within a specified mixing depth.
CAG:	Carcinogen assessment group. EPA group that prepared qualitative and quantitative carcinogenic risk assessments.
Cancer:	A cellular tumor, the natural course of which is fatal. Cancer cells, unlike benign cells, exhibit properties of invasion and metastasis (malignancy). Cancers are divided into two broad categories: carcinoma and sarcoma.
Carcinogenicity:	The extent to which a substance is able to induce a cancer response.
Catalyst:	A substance that promotes a chemical reaction. In the context of this report, a device installed on the tailpipe of a motor vehicle to control exhaust emissions.
CDM:	Climatological Display Model. A Gaussian dispersion model whose particular strength is its detailed area source

	treatment. Can also handle point sources, but not in as detailed a manner as ISC.
Centroid (population, source):	A single point whose coordinates represent the location of a BG/ED, in the case of a population centroid, or the location of an emission point, in the case of a complex source.
Chemical mass balance:	A type of receptor model, employing chemical methods for source impact determinations.
Chronic exposure:	Long-term exposure usually lasting six months to a lifetime.
Co-control:	In the context of air toxics, co-control represents the simultaneous control or mitigation of air toxics and criteria pollutants via the same control measure. For example, a motor vehicle catalyst would reduce VOC and CO emissions and also reduce benzene and other gaseous toxics.
Comparative potency factor:	A cancer unit risk factor for a complex substance or mixture that is extrapolated from human risk data for a reference substance based on the ratio of short-term bioassay responses of the complex substance to the reference substance. EPA is developing comparative potency factors for various sources of POM.
Complex facility:	A point source covering a large area and comprised of many, generally different, kinds of emission points such as fugitive equipment leaks, vents, stacks, volume sources, etc.
Complex terrain:	Terrain exceeding the height of a stack.
Composite plume:	The result of merging of multiple plumes downwind of an industrial complex.
Coverage (pollutant, source, spatial):	The extent of inclusion of pollutants or sources in an emission inventory or risk analysis or the amount of space or area represented in a monitoring program or inventory or risk analysis.
Cr+6:	Hexavalent chromium, i.e., chromium in the +6 valence state.
Criteria pollutants:	Pollutants defined pursuant to Section 108 of the Clean Air Act and for which national ambient air quality standards are prescribed. Current criteria pollutants include particulate matter, SO_x, NO_x, ozone, CO, and lead.

Culpability:	The extent to which something (generally a pollutant or source) is responsible for some effect (such as exposure or risk).
Decay:	A term that represents pollutant removal by physical or chemical processes.
Deposition:	The removal of particulate matter and gases, at a land or water body surface, by precipitation or dry removal mechanisms, including surface reactions and filtering.
Detection limits:	See minimum detection limits.
Dispersion coefficients:	Parameters used in Gaussian dispersion modeling to estimate plume growth through dispersion along the horizontal and vertical axes. These are computed as a function of downwind distance and atmospheric stability.
Dispersion modeling:	A means of estimating ambient concentrations at locations (receptors) downwind of a source, or an array of sources, based on emission rates, release specifications, and meteorological factors such as wind speed, wind direction, atmospheric stability, mixing height, and ambient temperature.
Dose-response assessment:	The determination of the relation between the magnitude of exposure and the probability of occurrence of the health effects in question.
DNPH:	Dinitrophenylhydrazine. Material used in special cartridges for monitoring formaldehyde and other aldehydes.
Effective stack height:	The height above ground level of the centerline of a plume. It is the sum of the physical stack height, plume, and stack-tip downwash (as applicable).
Excess cancer risk:	An increased risk of cancer above the normal background.
Exposure:	An event in which an organism comes into contact with a chemical or physical agent.
Exposure assessment:	Measurement or estimation of the magnitude, frequency, duration, and route of exposure to substances in the environment. The exposure assessment also describes the nature of exposure and the size and nature of the exposed populations.
Factor analysis:	A type of receptor model, employing chemical methods for source impact determination.

Fenceline:	A term used to represent the property boundary of a facility.
Fine particulate:	Particulate matter less than 2.5 microns in size.
FMVCP:	Federal Motor Vehicle Control Program. EPA's program to control motor vehicle emissions.
Formulation:	(see Model formulation)
Fugitive emission/release:	Emissions unconfined to a stack or duct, such as equipment leaks from valves, flanges, etc., or open spills.
GAMS:	GEMS Atmospheric Modeling System. GAMS is an Exposure model, similar to HEM/SHEAR, developed by EPA's Office of Toxic Substances.
Gaussian model:	A Gaussian dispersion model represents the distribution of concentrations within a plume by assuming a normal distribution along the horizontal and vertical axes. In the basic form, predicted concentrations are estimated as a function of emission rate, horizontal and vertical dispersion coefficients, and vertical and horizontal distance from the plume centerline.
GC:	Gas chromatography. A technique for separating compounds on a chromatographic column for subsequent analysis.
GC/MS:	Gas chromatography coupled with mass spectrometry for analysis of compounds.
GEMS:	Graphical Exposure Modeling System. An interactive, multimedia information management system that contains physiochemical parameters, fate data, and multimedia exposure models (e.g., GAMS), developed by EPA's Office of Toxic Substances.
Global buildup:	The widespread accumulation of pollutants in the atmosphere over the years. Global buildup is generally associated with more inert compounds such as halogens (e.g., carbon tetrachloride).
Grid:	A network of rectilinear or polar grid cells superimposed over an area, generally for modeling analyses. A rectilinear grid is defined by a series of perpendicular lines defining rectangular or square grid cells, whereas a polar grid is defined by a series of concentric circles and straight lines radiating from the center of the circles.

Grid cell:	The smallest area resolved within a modeling grid.
Grid spacing:	The dimensions of the grid cells within a grid.
Grid square:	A grid cell whose sides are equal.
Hazard identification:	The determination of whether a particular chemical is or is not casually linked to particular health effects.
HEM:	Human Exposure Model. EPA model used for exposure and risk analysis, which defines polar receptor grids around each point source. Can also model area sources by apportioning county level emissions to each BG/ED and running a simple box model. HEM contains two component modules: SHED and SHEAR. (Note: HEM is being upgraded by EPA.)
Highway vehicle source:	Car, truck, or motorcycle. Also called road vehicles or motor vehicles.
Hot spot:	A particular receptor, grid cell, or localized area wherein exposure or risk is high.
HPLC:	High performance liquid chromatography.
Hybrid modeling:	The use of several different models, particularly the mixed use of dispersion and receptor modeling, for complementary analyses.
IACP:	Integrated Air Cancer Program. EPA long-term research and development program to develop methods and conduct field and lab tests to learn what causes cancer in complex urban air mixtures and what sources are contributing to this cancer burden.
IEMP:	Integrated Environmental Management Project. A series of studies conducted in Philadelphia, Baltimore, Santa Clara, Kanawha Valley, and Denver to evaluate multimedia contributions to various health risks, with emphasis on cancer.
Incidence:	The frequency of occurrence of a certain event or conditions, such as the number of new cases of a specific disease or tumor occurring during a certain period. The incidence rate is the number of new cases during a certain period divided by the population size (e.g., 10 cases per 100,000 exposed persons).
Individual risk:	The increased risk for a person exposed to a specific concentration of a toxicant. May be expressed as a lifetime

	individual risk or as an annual individual risk, the latter usually computed as 1/70 of the lifetime risk.
IRIS:	Integrated Risk Information System. EPA computer system containing risk information (e.g., cancer unit risk factors) for specific chemicals.
ISC:	Industrial Source Complex model. EPA Gaussian dispersion model designed to handle complex sources. ISC contains a long-term module (ISCLT) and a short-term module (ISCST).
Lifetime:	Considered to be 70 years in EPA health risk assessments.
LONGZ:	A predecessor model of ISCLT, containing many of the source-specific features of ISCLT. A long-term (seasonal or annual average) Gaussian dispersion model that is used for point and urban-wide area sources. This model can accept site-specific turbulence data to estimate local dispersion rates.
MADAM:	Monitoring of Ambient Data Assessment Module. Module within EPA's PIPQUIC system used to evaluate model performance by comparing measured and model predicted ambient air quality data and partitioning the results by various meteorological parameters.
MEI:	Maximum exposed individual.
Microenvironment:	Localized environment in which one may be exposed to pollutant concentrations that differ considerably from ambient (outdoor) air (e.g., indoor household air, occupational exposures, air within automobiles, etc.) EPA's TEAM studies evaluate personal exposures as individuals are exposed to air in different microenvironments during each day.
Micron:	One millionth of a meter. A dimensional unit used to measure the diameters of particles.
Microscale:	The immediate vicinity of a source, e.g., within 1 to 2 kilometers.
Minimum detection limit:	The lowest level measurable by a monitoring technique at some level of confidence.
MIR:	Maximum individual risk, i.e., risk to the most exposed individual.

Mitigation:	The reduction or control of emissions, exposures, or risks due to air toxics.
Mixing height: (Or mixing depth)	The height above the surface through which vertical mixing occurs without suppression by an elevated stable layer.
Mobile source:	Any motorized vehicle, such as cars, trucks, airplanes, or trains. Sometimes refers specifically to highway vehicle sources.
Model formulation:	A model formulation is determined by the model selected, the specific input data, and options selected for a model run. A range of model formulations could be made using the same dispersion model.
Model performance:	The evaluation of the performance of a dispersion model by comparing modeled concentrations to measured air quality data, generally based on statistical tests such as measures of bias, variance, and correlation.
Model(ing):	See dispersion modeling or receptor modeling. This term can also be used in the context of emissions modeling, referring to the prediction of emissions from a source or source category.
Modeling protocol:	As used in this report, a modeling protocol provides a detailed account of the specific model formulation(s) that will be used to perform a dispersion modeling analysis. It generally is used to obtain comment prior to doing a detailed analysis.
Monitoring:	The collection and analysis of ambient air samples. Sometimes refers specifically to sampling alone and not to analysis. Can also refer to source (stack) sampling.
Motor vehicle:	On-road or off-road cars, trucks, or motorcycles.
Multiple linear regression analysis:	A type of receptor model employing chemical methods for source impact determination.
Mutagenicity:	The extent to which a chemical or physical agent interacts with DNA to cause a permanent, transmissible change in the genetic material of a cell.
NAAQS:	National Ambient Air Quality Standard. Set by EPA for criteria pollutants under the Clean Air Act.

NEDS:	National Emissions Data System. EPA's centralized emission inventory of criteria pollutant emissions.
NEM:	NAAQS Emissions Model. EPA exposure model that considers movement of individuals through various microenvironments.
NESHAP:	National Emission Standard for Hazardous Air Pollutant. Standards set by EPA for hazardous air pollutants under Section 112 of the Clean Air Act.
Network:	An array of ambient air monitors distributed over an area.
Network enhancement:	The enhancement of an ambient air network to improve spatial or temporal coverage. Sometimes done on a temporary basis.
Noncancer risk:	Risk of a health effect other than cancer.
Nontraditional sources:	Sources not usually included in an emission inventory, such as wastewater treatment plants, ground-water aeration facilities, hazardous waste combustors, landfills, etc., which are air emitters due to intermedia transfer from water or solid waste.
Normalized modeling:	Modeling of unit weights (e.g., 1 mg/yr) of emissions from each source, rather than modeling of actual emissions, and displaying incremental receptor concentrations or receptor coefficients. Thereafter, the resulting normalized receptor coefficients are adjusted by actual emission rates to simulate different emission scenarios rather than re-running the model over and over with different emissions totals. This process assumes linearity between emissions and modeled ambient air concentrations, which does not always hold true if stack and exhaust parameters change.
NSR:	New Source Review. Permit process for evaluating emissions and need for controls before construction and operation of a proposed facility. EPA, as well as many States and local agencies, has NSR requirements for air toxics sources.
OPPE:	EPA's Office of Policy, Planning, and Evaluation. Initiator of the Integrated Environmental Management Project, a series of geographic, multimedia studies in various cities. (See IEMP.)
PAN:	Peroxyacetyl Nitrate. A photochemical oxidant formed in urban atmospheres along with ozone.

Personal Monitoring:	Sampling done in EPA's TEAM study by individuals wearing personal monitors.
Photochemically formed pollutant:	A secondarily formed pollutant due to atmospheric photochemistry, e.g., formaldehyde or PAN.
PIC:	Products of incomplete combustion. A term used somewhat loosely in various studies referring generally to polycyclic organic matter.
PIPQUIC:	Program Integration Project—Queries Using Interactive Commands. A data handling system developed by EPA as part of the IEMP to calculate exposures and risks from air toxics emissions data. PIPQUIC is being used in various urban air toxics studies.
PM:	Particulate matter.
Point source:	A source large enough for individual record to be kept in an emission inventory, often emitting above a certain cutoff level or threshold.
Polar grid:	See grid.
POM:	Polycyclic organic matter. A broad class of compounds that generally includes all organic structures having two or more fused aromatic rings (i.e., rings sharing a common border). POM includes polynuclear aromatic hydrocarbons (PAH or PNA).
Population risk:	Generally synonymous with areawide incidence.
Primary pollutant:	One emitted directly from an emission source prior to any secondary physical or chemical reaction.
PUF:	Polyurethane foam. An adsorbent material used for sampling semivolatile organic compounds.
QA/QC:	Quality assurance/quality control.
Receptor:	A particular point in space where a monitor is located or where an exposure or risk is modeled.
Receptor grid:	An array of receptors. Generally synonymous with network.
Receptor modeling:	A technique for inferring source culpability at a receptor(s) by analysis of the ambient sample composition. There are

	various receptor models employing microscopic and chemical methods for analysis.
Release specifications:	Used as model inputs to characterize the location, release height, and buoyant and momentum fluxes of each source. Required terms include stack height, exit velocity, inner stack diameter, exhaust temperature, and the dimensions of nearby structures.
Risk:	The probability of injury, disease, or death under specific circumstances. In quantitative terms, risk is expressed in values ranging from zero (representing the certainty that harm will not occur) to one (representing the certainty that harm will occur).
Risk assessment:	The use of the factual base to define the health effects of exposure of individuals or populations to hazardous materials and situations. May contain some or all of the following four steps:
Risk characterization:	The description of the nature and often the magnitude of human risk, including attendant uncertainty.
Risk management:	The decision-making process that uses the results of risk assessment to produce a decision about environmental action. Risk management includes consideration of technical, social, economic, and political information.
Road vehicle source:	See highway vehicle source.
Sample compositing:	The combining of samples before analysis to increase the temporal or spatial representativeness, while reducing analytical costs.
Sampling:	See monitoring or ambient air monitoring
Sampling duration:	The length of time (generally in hours) each sample is taken or drawn (e.g., 12 or 24 hours).
Sampling frequency:	The length of time between samples (1 hour, 1 day, 6 days, etc.).
Sampling period:	The length of time (days, months, years) for which a sampling program is operational.
SCAQMD:	South Coast Air Quality Management District. The local air pollution control agency in California responsible for the Los Angeles area.

Scoping study:	Also known as a screening study. An assessment or analysis using tentative or preliminary data whose results are not accepted as absolute indicators of risk or exposures, but rather are taken as an indication of the relative importance of various sources, pollutants, and control measures. Most urban air toxics assessments conducted to date have been considered to be scoping studies, useful for pointing out where more detailed work is needed prior to regulation.
SCREAM:	South Coast Risk and Exposure Model. An enhancement of EPA's HEM/SHEAR developed by SCAQMD that uses more detailed population and meteorological data.
Screening study:	See scoping study.
Secondary pollutant:	Also, "secondarily formed pollutant." A pollutant formed in the atmosphere as a result of chemical reaction and/or condensation, such as PAN. Some pollutants (e.g., formaldehyde) are both primary and secondary pollutants.
Semivolatile organics:	Compounds that have vapor pressures (in clean air) of 10^{-8} to 10^{-4} torr, and which readily adsorb upon particulate matter. Not clearly gaseous or particulate under all conditions.
SHEAR:	Systems Application Human Exposure and Risk. A module within EPA's Human Exposure Model designed to focus on multiple pollutant, multiple source exposures, including area source analyses. SHEAR uses a Gaussian dispersion model for point sources and a box model for area sources.
SHORTZ:	Short-term version (e.g., 1-hr to 24-hr averaging periods) of SHORTZ/LONGZ companion models. See LONGZ.
SIC:	Standard Industrial Classification. A series of codes or classifications to categorize industry, published regularly by the Office of Management and Budget.
SIP:	State Implementation Plan. Required by States under the Clean Air Act to indicate a plan of action to meet National Ambient Air Quality Standards for criteria pollutants.
Source apportionment:	See receptor modeling.
Source grid:	A grid defined to encompass all emission sources that one wants to model. The source grid is sometimes defined bigger than a corresponding receptor grid so that all local sources impacting on the receptor grid will be considered.

	More typically, the source grid and receptor grid coincide in most studies.
Spatial coverage:	The area included or covered by a sampling network or a source/receptor grid.
Spatial resolution:	The extent to which emissions, monitoring or any other data are subdivided or resolved in space, generally across a geographical area. For example, emissions data may be spatially resolved to 1 kilometer by 1 kilometer squares within an urban area.
Species profile:	A set of apportioning factors that allow one to subdivide VOC or PM emission totals into individual chemicals or chemical classes.
Stability:	A parameter to describe the degree of turbulence in the atmosphere, ranging from unstable (vigorous mixing) to stable (suppressed mixing).
Subchronic exposure:	Exposure to a substance spanning approximately 10 percent of the lifetime of an organism.
Supersite:	A monitoring site that alone, or in conjunction with other sites, best represents the scale of interest, such as suburban neighborhoods, central business district, or rural areas. Such sites can be inferred by statistical analysis of modeled data.
Surrogate indicator:	A variable whose spatial or temporal distribution is assumed to behave in the same manner as some variable of interest. Surrogate indicators are used for spatial and temporal apportionment of emissions data, especially for area sources.
TEAM:	Total Exposure Assessment Monitoring. The type of monitoring being conducted by EPA's Office of Research and Development to measure total human exposures of individuals as they occupy various microenvironments, such as outdoors, indoors, and commuting in motor vehicles.
Temporal resolution:	The extent to which some variable, typically emissions and monitoring data, is subdivided or resolved in time. Data, for example, may be resolved hourly or seasonally.
Tenax:	A porous polymer material often used for sample collection of certain organic materials.

Tracer:	Tracer pollutants are used in receptor modeling to estimate the contribution of a source, or source category, to total ambient levels of the pollutant or pollutant class (such as PM-10) of interest. Ideally, tracers are unique to a source or source category.
Transformation:	The conversion, through chemical or physical processes, of one compound or several compounds into other compounds as a result of aging and irradiation in the atmosphere.
Transport:	The movement of pollutants by wind flow. Transport is characterized for modeling purposes by wind speed and wind direction.
UNAMAP model(s):	User's Network for Applied Modeling of Air Pollution. A set of dispersion models compiled by EPA that is used to support regulatory and other needs for modeled data.
Unit cancer risk factors:	The incremental upper bound lifetime risk estimated to result from a lifetime exposure to an agent if it is in the air at a concentration of 1 microgram per cubic meter.
Urban soup:	An expression referring to the multi-source, multi-pollutant urban air toxics problem resulting from the complex interaction of many pollutants, sources, and atmospheric transformation.
VMT:	Vehicle miles traveled. Mobile source emission factors are typically expressed in terms of grams per VMT.
VOC:	Volatile organic compounds.

APPENDIX B

EVALUATION OF OPTIMUM SIZE AND LOCATION OF AIR TOXICS MONITORING NETWORKS BASED ON SPATIAL CORRELATIONS OF CONCENTRATION

Introduction

The purpose of this Appendix is to describe a method to select optimum monitoring sites based on dispersion modeling and statistical analysis. This method uses modelled emissions data to design a monitoring network that best meets project objectives within a specific study area. The ultimate goal of this methodology is to select a limited number of monitoring sites that yield relatively independent (i.e., noncorrelated) air quality data, and to avoid selecting sites that provide relatively little new information. Appendix B is organized to first describe this optimization method, and then to provide a limited evaluation of its effectiveness based on the IEMP Philadelphia measured air toxics data set. This measured data set is used to evaluate how well the modeling-based site selection approach corresponds with the selection of optimal sites ("Supersites").

B.1 Method

There are six steps, which lead to a quantitative determination of the sites that best meet study objectives based on the best available emissions data. The steps can be briefly summarized as follows.

Step 1 - Select a "long list" of candidate monitoring sites. Select a relatively broad range of candidate monitoring sites. In many metropolitan areas this set could include all existing criteria pollutant monitoring sites and other readily accessible locations, such as schools, state/county facilities, etc. The goal of Step 1 could be to provide broad coverage of candidate sites throughout the study area.

Step 2 - Compile emissions data. Obtain (or compile as necessary) available emissions data for point and area sources (such as mobile sources, degreasing, etc.)

Step 3 - Perform dispersion modeling analysis. Perform dispersion modeling using emissions data obtained in Step 2. The meteorological data ideally would be five[1] separate years of sequential data to represent the season(s) proposed for the monitoring program. The averaging period should also match that proposed for the monitoring program, e.g., 24-hour periods. If every 3rd of 6th day sample frequencies will be used for the monitoring program, the same should be done for the modeling. In short, the goal of the modeling is to compile an output data set comparable to the measured data set.

Step 4 - Perform statistical analysis. Correlation matrices for each pollutant are compiled to show "R" values for all site combinations. The correlation considers all "samples." For example, if a three-month sampling period is proposed, taking 24-hour integrated samples on every third day, then there would be 30 modeled 24-hour averages for each site and each pollutant.

Step 5 - Group sites. Group sites by combining sites correlated by 0.7 or higher into one cluster. (A 0.7 correlation coefficient was arbitrarily selected. A higher or lower value could be selected that best met project needs.) There would be separate groups of sites for each pollutant. Assign a score of +1 to each cluster and subdivide the score among the members of the cluster. This is done separately for each pollutant, and then the scores at each site are summed across all pollutants.

Step 6 - Successive elimination of sites. The site with the lowest score is eliminated and scores recomputed (as was initially done in Step 5). Another site is eliminated as above and the scores recomputed, and so forth, until a core network (e.g., 3 sites) is described.

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[1] If multiple years of meteorological data are obtained, the model analysis is run for each receptor for each year to assess the influence of meteorological variability on site selection.

B.2 Example Application: Philadelphia IEMP

This example application of the supersite concept first shows site selection based on modeled data, followed, as a comparison, by sites that would be selected based on the actual measured data set. Within the measured data set greater detail is provided in the site elimination procedures of Step 6. (The reader is referenced to the model performance study conducted for the IEMP Philadelphia Study [2] for background information on the measured data set used for this example.)

Siting Analysis Based on Modeled Data

The IEMP Philadelphia Study operated a ten-site monitoring network[3] during the period of 1983-1984. Using these ten locations as "candidate sites," dispersion modeling was used to select sites expected to be most independent. Independence was defined as having a correlation (R value) of less than or equal to 0.70 with all other "candidate sites."

For benzene, ethylbenzene, trichloroethylene, and xylene all correlations in Step 1 were above the cutoff of 0.7 (in fact, they were above 0.84, 0.70, 0.73, and 0.79, respectively), making all ten sites into one cluster. This is attributed to the dominance of area sources in the model and the inability to detect many of the spatial variabilities caused primarily by mobile sources and possibly by gasoline marketing. The remaining clusters are:

Carbon tetrachloride	3,5, and 10; 4,7, and 10; 3,4,5,8, and 9
Chloroform	2 and 9; 3,4,5,7,8,9, and 10
Ethyl-chloride	2 and 3; 5,,7,8, and 10; 5,8,9, and 10
Perchloroethylen	1,2,3, and 5; 3,4,5,7,8,9, and 10
1,2-dichloroethane	2 and 3; 5,7,8, and 10; 8 and 9

Elimination of sites by Steps 5 and 6 of the site selection procedure leads to:

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[2] Sullivan, D. A., 1985 . Evaluation of the Performance of the Dispersion Model SHORTZ for Predicting Concentrations of Air Toxics in the U.S. Environmental Protection Agency's Philadelphia Geographic Study. U.S. Environmental Protection Agency, Integrated Environmental Management Divisions, Washington, D.C.

[3] Nine sites had high enough data recovery to support the objectives of this analysis.

Step	Site Eliminated	Sites Left
Stage 1	7 out:	1,2,3,4,5,8,9,10
Stage 2	5 out:	1,2,3,4,8,9,10
Stage 3	8 out:	1,2,3,4,9,10
Stage 4	3 out:	1,2,4,9,10
Stage 5	10 out:	1,2,4,9
Stage 6	4 out:	1,2,9
etc.		

This process would thus prescribe sites 1, 2 and 9 as yielding the most independent information, and thus represent the optimum location for 3 monitoring sites. The process could be truncated or continued to yield any number of sites, down to a single location.

Siting Analysis Based on Measured Data

As a test of the site selection approach described above, the measured data set from the Philadelphia IEMP air toxics monitoring network was used in place of the modeled data used in the previous subsections. The goal was to use the measured data set to help evaluate the usefulness of the model-based siting procedure. (The following discussion describes the site elimination process in detail as it was carried out in Phildelphia.)

The final clusters of Step 1 based on measured data are as follows:

Compound	Clusters	Number of Sites Needed for Network[4]
Benzene	9 and 10	8
Carbon Tetrachloride	4 and 9	8
Chloroform	1,3,4,8,9, and 10	4
Ethylbenzene	1 and 5; 4,7, and 8	6
1,2-Dichloroethane	1 and 10	8
Perchloroethylene	4 and 10; 8 and 10	7
Toluene	Sample size too small	–
Trichloroethylene	3,4, and 8; 9 and 10	6
Xylene	4 and 8; 7,9, and 10	6
1,2-Dichloropropane	1 and 4; 3 and 5	7

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[4] Considering one compound at a time.

We note that for perchloroethylene, site 10 is associated with sites 4 and 8, although 4 and 8 are not highly correlated (0.58). We then assigned a score of +1 to each cluster and subdivided the score among the members of each cluster. Adding up the scores across the nine chemicals produced the following values:

Site	Score
1	6.67
2	9.00
3	7.00
4	4.84
5	8.00
7	7.67
8	5.84
9	5.84
10	5.84

From this analysis, site 4 clearly produced the least information, i.e., for seven of the nine compounds considered, site 4 was well correlated with other site(s) at the 0.7 level or higher, and added marginal information to the network. Consequently, if one wanted to have a network of only nine sites, site 4 would be the first one to eliminate based on this approach.

Next we assumed that station 4 was eliminated and repeated the above analysis. This resulted in the information scores:

Site	Score
1	7.20
2	9.00
3	7.20
5	8.00
7	7.83
8	6.70
9	6.53
10	5.53

Thus, the next site we would eliminate would be site 10. Continuing in this manner we created networks with fewer and fewer sites until only three sites remained. At that stage, the sites were all independent and so these represented a "core". The networks we obtained via this procedure are as follows:

		Remaining Sites
Stage 1	4 out:	1,2,3,5,7,8,9,10
Stage 2	10 out:	1,2,3,5,7,8,9
Stage 3	8 out:	1,2,3,5,7,9
Stage 4	3 out:	1,2,5,7,9
Stage 5	9 out:	1,2,5,7
Stage 6	1 out:	2,5,7

It should be noted that based on the initial "value" ranking that sites 2,5, and 7 had the highest scores: Site 2 (9.0), site 5 (8.0) and site 7 (7.7). This result is not surprising when one considers the orientation of sources along the corridor in which the monitoring network is located. Site 2 is located in the downtown area, which is substantially different than the locations of the other sites that are in more industrialed areas. Site 5 is located in the heart of the industrial area of Bridesburg, and as such would be expected to be relatively independent. Site 7 is located in an area north of the rest of the network, and experienced substantially elevated concentrations for automobile related emissions with flow from the southwest, and as such, could be expected to be relatively independent based on the measured data.

Method Evaluation

We note that at the end of the fourth stage of the modeling analysis, we have five sites (1,2,4,9 and 10) in our proposed network, which include at least one representative from each cluster for each of the nine chemicals. In contrast, sites 1, 2, 5, 7 and 9 would have been selected as optimum based on the actual measured data set. It is interesting to note that sites 5 and 7 were preferred sites based on the measured data while those same two sites were the first to be eliminated for the theoretical model data. On the other hand, the model-based process selected sites 1,2, and 9 as important; these sites were among the five most informative for the actual data. Therefore, it appears that there is some value in using dispersion models and spatial correlation analysis as a guide to site selection for air quality studies. Caution should be used in selecting only the top few sites, because as shown, the modeled data did well in selecting the top five sites, but resulted in significant differences compared to the measured results when selecting a 2-3 site network. Further data needs to be evaluated to fully evaluate this approach.